"STRAW MAN" BASELINE PROBLEM FORMULATION

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GLOSSARY

Management Goal: A general statement, usually in legislation or regulation, such as "restore and maintain the chemical, physical, and biological integrity of the waters of the Great Lakes Ecosystem." (USEPA 2001)

Management Objective: A specific statement about a desired condition or direction such as, "meet Wisconsin Water Quality goals and objectives." (USEPA 2001)

Assessment Endpoint: An explicit expression of what is to be protected such as, "survival, growth and reproduction of wildlife must be such that wildlife populations are maintained at sustainable levels." (USEPA 1997).

Measurement Endpoint: A measurable environmental characteristic that is related to the assessment endpoint. For example, "the mean concentration at a selected downstream location based on a biweekly sampling frequency meets the most stringent water quality criteria." (USEPA 1997).

Decision Statement: Essentially an "If....then" statement that indicates at which point remedial action will be implemented. It can reference a cleanup level or action level (see below). (USEPA 2000)

Action Level: The criterion for choosing among alternative remedial actions, natural recovery versus dredging and disposal. (USEPA 2000)

Decision Rule: The determination of what statistical parameter of the potentially affected receptor population must exceed the action level for the alternative action, i.e. remediation, to be implemented. (USEPA 2000)

Remedial Action Outcome (RAO): A general description of what the cleanup is expected to accomplish, and helps focus the development of the remedial alternatives in the feasibility study. (USEPA 2002).

Remediation Goals (RGs) or Preliminary Remediation Goals (PRGs): Criteria or risk-based levels that are considered protective of human health and the environment which are identified early in the remedial investigation process (at scoping meetings) based on readily available information [e.g., from the PA/SI (preliminary assessment/site inspection), NPL (National Priorities List) listing packages, or screening risk assessments]. As more information is generated during the RI, these PRGs may be modified to incorporate an improved understanding of site conditions, resource use, human activities, and the nature and extent of contamination (USEPA 2002).

Cleanup Level: At most sites, RGs are developed into final, chemical-specific, sediment cleanup levels by weighing the NCP balancing and modifying criteria and other factors relating to uncertainty, exposure, and technical feasibility (USEPA 2002).

1.0 INTRODUCTION

The term "straw man" in the sense that it is used here is: "a document that temporarily stands in for and is eventually replaced by something more substantial." This "straw man" is provided as an illustration of how Xcel Energy believes EPA guidance for ecological risk assessment should be synthesized with recent EPA strategy and guidance for management of contaminated sediment sites to develop a systematic and objective basis for addressing issues in the sediment Operable Unit of the Ashland, Wisconsin Lakefront Superfund Site (Ashland). This "straw man" will follow the framework of a baseline problem formulation and will:

- Propose a preliminary conceptual site model which incorporates a consideration of sediment stability;
- Make recommendations for the risk management goal and associated risk management objectives for the Ashland site;
- Make recommendations for assessment endpoints which Xcel Energy believes are the critical ones that will influence Ashland site management decisions;
- Develop risk questions and risk hypotheses associated with these assessment endpoints;
- Propose measurement endpoints that can be used to address risk hypotheses;
- Present a data gap analysis based upon these endpoints and risk hypotheses;
- Recommend approaches to acquire the necessary data to address the risk hypotheses;
- Discuss how the data quality objective (DQO) process should be used to develop the detailed study plans for measurement endpoints; and
- Discuss how the DQO process should be used to identify how management decisions for Ashland site sediments will be related to the outcome of risk assessment.

The integration of these important elements of the risk assessment problem formulation, sediment management guidance and DQO process is illustrated in Figure 1.

The Guidance that we have followed to develop this straw man Baseline Problem Formulation includes:

Ecological Risk Assessment for Superfund: Process for designing and conducting ecological risk assessments, Interim Final. Environmental Response Team, Edison, NJ. (USEPA 1997);
Guidance for Ecological Risk Assessment. (USEPA 1998);
Planning for Ecological Risk Assessment: Developing Management Objectives. (USEPA 2001);
Guidance for the Data Quality Objective Process (USEPA 2000);
Principles for Managing Contaminated Sediment Risks at Hazardous Waste Sites. OSWER Directive 9285.6-08; and,
DRAFT Contaminated Sediment Remediation Guidance for Hazardous Waste Sites. OSWER 9355.0-85. (USEPA 2002).

Xcel Energy believes that there is good advice available to guide us in addressing the potential issues associated with contaminated sediments at the Ashland site and to agree on a risk management remedy that is protective of human health and the environment. However, Xcel Energy also believes that the groundwork for realizing this objective needs to be thoroughly developed at the front end of the process and must involve all potential Interested Parties. The following, from *Critical Issues for Contaminated Sediment Management, MESO-02-TM-01* (Apitz et *al.* 2002), stresses the importance of reaching agreement early in the process on how the results of the risk assessment should be related to remedial action outcomes (RAOs),

> "Data are frequently collected without input or consensus from the full suite of stakeholders. A stumbling block to efficient data use has been the inability to compile and integrate or synthesize data, especially from multiple sources, in a uniform or mutually agreeable manner. Another problem at times has been a lack of openness in the process, resulting in distrust and extreme positions by some regulators, RPMs and stakeholders. Unless these issues are resolved up front, progress is unlikely at a site. One of the major observations of the National Research Council (1997) was that successful case studies resulted from openness, communication, and buy-in by all parties early in the process [....].
>
> Ecological risk assessment and environmental management, [.........], are complex, multivariate and uncertain processes. The goal of a well-designed RI/FS is to collect and evaluate enough information to reduce uncertainty and increase the probability that risk is properly assigned and managed. However, in most cases, "truth" is not clear-cut, and some level of uncertainty remains. For example, it is rare that a direct correlation between multiple lines of evidence (i.e., toxicity and bulk COC [contaminants of concern] concentration) is observed, due to multiple confounding factors. This does not obviate the utility of each measurement, but argues for a weight-of-evidence approach. <u>Unless, however, decision criteria are negotiated up front, preferably before data are collected</u> (emphasis added), there can be a temptation by parties to exploit uncertainty in either direction to avoid "undesirable" outcomes. Clearly negotiated decision frameworks, adherence to DQOs [data quality objectives], and an explicitly defined Site Conceptual Model [.........] makes it easier for all parties to have ownership of the process and to negotiate in good faith."

A more recent report by the National Research Council entitled, *"A Risk Management Strategy for PCB Contaminated Sediments"*, (NRC 2001) emphasized this concept and made several recommendations, many of which were incorporated into the 11 *"Principles for Managing Contaminated Sediment Risks at Hazardous Waste Sites"* (OSWER 2002). Several of these principles are ones that should be addressed in the initial stages of site investigation and require input from a risk assessment perspective to ensure that the results of the remedial investigation will provide an adequate foundation for a risk assessment.

One unique aspect of the risk assessment framework is that it can be an iterative process and can thereby benefit from work and planning done previously. This is the basis for Principle # 5 (OSWER 2002) and the NRC recommendation that, "The [risk] framework is intended to supplement, and not supplant the CERCLA remedial process mandated by law for Superfund Sites." It is therefore expected that any future site investigation will benefit from an iterative re-evaluation of the adequacy of the basis for remedial and risk management decision-making.

Although there have been a number of remedial investigation studies as well as two previous ecological risk assessments competed at the Ashland site, Xcel Energy believes that the ongoing remedial investigation process and the future risk management decision-making will benefit <u>now</u> from a more formal and systematic integration of concepts introduced in the guidance for managing contaminated sediment sites (USEPA 2002) into the risk assessment process. Xcel Energy hopes that this will serve as an opportunity to re-evaluate the adequacy of the foundation for remedial decision-making as well as ensuring that there will be a clear and transparent relationship between proposed remedial action outcomes and risk management goals.

As indicated above, the following Baseline Problem Formulation is intended to serve as an example and as such is not comprehensive. It emphasizes how select measurement endpoints and risk hypotheses are related to management objectives and management goals and how they lead to developing an analysis plan that provides the requisite information for supporting risk management decisions on RAOs. In selecting candidate management goals, management objectives and assessment endpoints, however, Xcel Energy has proposed those that we believe will be the critical ones necessary to support risk management decision making for the Ashland site Sediment Operable Unit.

2.0 STEP 3: BASELINE PROBEM FORMULATION

This Straw Man Baseline Problem Formulation assumes that in addition to the traditional objectives of Baseline Problem Formulation discussed in the following text box (emphasis added) from EPA guidance (USEPA 1997) additional objectives reflecting the relevant Principles for Managing Contaminated Sediment Risks at Hazardous Waste Sites (OSWER 2002) will be addressed.

OVERVIEW

Step 3 of the eight-step process initiates the problem-formulation phase of the baseline ecological risk assessment. Step 3 refines the screening-level problem formulation and, with input from stakeholders and other involved parties, expands on the ecological issues that are of concern at the particular site. In the screening-level assessment, conservative assumptions were used where site-specific information was lacking. In Step 3, the results of the screening assessment and additional site-specific information are used to determine the scope and goals of the baseline ecological risk assessment. Steps 3 through 7 are required only for sites for which the screening-level assessment indicated a need for further ecological risk evaluation.

Problem formulation at Step 3 includes several activities:

- Refining preliminary contaminants of ecological concern;
- Further characterizing ecological effects of contaminants;
- Reviewing and refining information on contaminant fate and transport, complete exposure pathways, and ecosystems potentially at risk;
- Selecting assessment endpoints; and
- Developing a conceptual model with working hypotheses or questions that the site investigation will address.

At the conclusion of Step 3, there is a SMDP, which consists of agreement on four items: the assessment endpoints, the exposure pathways, the risk questions, and conceptual model integrating these components. The products of Step 3 are used to select measurement endpoints and to develop the ecological risk assessment work plan (WP) and sampling and analysis plan (SAP) for the site in Step 4. Steps 3 and 4 are, effectively, the data quality objective (DQO) process for the baseline ecological risk assessment.

These additional objectives include:

Principle 4: Develop and Refine a Conceptual Site Model that Considers Sediment Stability
Principle 8: Ensure that Sediment Cleanup Levels are Clearly Tied to Risk Management Goals

In addition, it is recommended that the full involvement of all Interested Parties in the Baseline Problem Formulation process be solicited. Involvement of all Interested Parties, including the *risk managers* and other *stakeholders,* during this phase and often throughout the rest of the risk assessment is encouraged to ensure that the results of the risk assessment can be used to support management decisions, and that the risk assessment and risk management results are effectively communicated. The formal inclusion of other Interested Parties satisfies two other Principles:

Principle 2: Involve the Community Early and Often
Principle 3: Coordinate with States, Local Governments, Tribes and Natural Resource Trustees

While risk management decisions should be independent from the risk assessment process, risk managers have a vested interest that adequate and appropriate information be collected during the remedial investigation to support the range of risk management decisions. The incorporation of Scientific-Management Decision Points (USEPA 1997) throughout the risk assessment process encourages this collaboration between risk managers and risk assessors. The involvement of risk managers during the Problem Formulation ensures that the assessment endpoints selected are compatible with management goals and management objectives (USEPA 2001), and that any sampling conducted for measurement endpoints is sufficient to unambiguously support risk management decisions.

The process of relating sediment cleanup goals to risk management goals takes place in the initial stages of the risk assessment problem formulation where risk assessors and risk managers collaborate to define the overall risk management goals for the site and establish a framework for deciding how the results of the risk assessment will be used to make risk management decisions (Figure 1). The process of integrating the Risk Assessment Problem Formulation (Step 3 in the Ecological Risk Assessment for Superfund (ERAGS) (USEPA 1997) and Data Quality Objective Process (Step 4 in ERAGS) is described in ERAGS,

> "At the conclusion of Step 3, there is a SMDP [Scientific-Management Decision Point], which consists of agreement on four items: the assessment endpoints, the exposure pathways, the risk questions, and conceptual model integrating these components. The products of Step 3 are used to select measurement endpoints and to develop the ecological risk assessment work plan (WP) and sampling and analysis plan (SAP) for the site in Step 4. Steps 3 and 4 are, effectively, the data quality objective (DQO) process for the baseline ecological risk assessment (USEPA 1997)."

The development of *management goals* and their related *management objectives* are the foundation for both the risk assessment as well as for development of RAOs and Preliminary Remediation Goals (PRGs) (USEPA 2002). As discussed in *Planning for Ecological Risk Assessment: Developing Management Objectives* (USEPA 2001), these management goals and objectives should be established prior to or as an initial element of the risk assessment problem formulation when management of a site is based upon risk. Until management goals are identified it is difficult to agree upon risk assessment endpoints or more importantly, *decision statements* and *action levels* relating to RAOs and PRGs. The DQO process provides a method for agreeing upon *action levels* and *decision statements* about implementation of appropriate remedial action and *decision rules* for risk management decisions (USEPA 2000). *Decision rules* provide a basis for agreeing upon how risk management decisions are related to *action levels* and upon the amount of uncertainty that is tolerable to the risk manager in making these risk management decisions.

However, risk management of some sites may not lend itself to establishing "bright-line" criteria that determine what risk management decision is appropriate given certain results of the risk assessment and some Interested Parties will not feel comfortable losing their flexibility to change perspective once risk assessment results are final. For this reason Xcel Energy has proposed a modified approach to achieving some *a priori* consensus on how the results of risk assessment relate to risk management outcomes. This approach is discussed in Section 2.6. This approach considers that if the risk manager for the Ashland site believes that establishing "bright-line" criteria for selecting RAOs, as would occur if the DQO process was fully utilized, is not achievable, then at least the DQO process can be used to focus the risk assessment on **only** those objectives that are the "drivers" for risk management decision-making.

2.1 Scope of Baseline Ecological Risk Assessment

It should be noted that the baseline ecological risk assessment (BERA) is not an impact statement, it does not attempt to document the degree of impact to reproduction, growth or survival but focuses primarily on whether exposure to site contaminants pose an unacceptable risk of adverse effects to any of the assessment endpoints developed as part of this problem formulation.

2.2 Nature & Extent of Contamination

The Ashland Lakefront Property that is the subject of this Problem Formulation is located in Section 33, Township 48 North, Range 4 West in Ashland County, Wisconsin. The property consists of a flat terrace adjacent to the Chequamegon Bay shoreline. It is bounded by Prentice Avenue and a jetty extension of Prentice Avenue to the east, the Ellis Avenue and the marina extension of Ellis Avenue to the west, Chequamegon Bay to the north, and the Wisconsin Central Limited railway to the south. The property is owned by the City of Ashland, and is currently being utilized as a park (Kreher Park); a portion of the property is occupied by the City's former wastewater treatment plant.[1]

The property has been the subject of numerous investigations. Previous investigations have identified fill at the Kreher Park portion of the property that ranges in thickness from 0 to 10 feet.

[1] The entire Ashland Lakefront site that is the subject of the CERCLA investigation includes not only the affected bay sediments and Kreher Park, but the upland properties on the Xcel Energy property south of the railroad.

The near shore sediments range in thickness from 0 to 9 feet. The fill material and lake sediments are underlain by the Miller Creek Formation. In the Ashland area, the Miller Creek has been identified up to 50 feet in thickness. At Kreher Park, the Miller Creek ranges in thickness from 6 to 20 feet, increasing in thickness from south to north. The Miller Creek consists predominantly of a dense silty clay till with inter-bedded lenses of silt, sand, and gravel. The Miller Creek is underlain by the Copper Falls Formation which consists of granular cohesionless material. An artesian well located on the marina jetty was installed in the Copper Falls at a depth of more than 100 feet. The entire thickness of the Copper Falls has not been penetrated as part of any site investigation with borings advanced at Kreher Park, or on the NSP/Xcel Energy property to the south; it is believed to be up to 200 feet thick. The Copper Falls is underlain by Precambrian sandstones of the Oconto Group; the uppermost bedrock unit.

Previous investigations have identified contamination in Kreher Park and in near shore bay sediments. Contaminated near shore sediments are located within the inlets created by the jetty and marina extension described above. Contaminants of potential concern (COPCs) identified during previous investigations of the sediments include VOCs and SVOCs characteristic of a coal tar/creosote origin. Naphthalene was detected as both a VOC and an SVOC (it is common to both test methods), and has been the most common constituent detected.

2.2.1 Contaminants of Potential Concern

Benzo(a)pyrene, benzo(a)anthracene, xylenes, ethylbenzene, and other VOCs have contaminated soils and underlying ground water, and have migrated to Chequamegon Bay.

(NOTE: To the extent possible the Baseline Problem Formulation should attempt to relate COPCs, if they are unique, to historical sources).

2.2.2 Refinement of Chemicals of Potential Concern

It is customary in the Baseline Problem Formulation to refine the list of contaminants for the BERA by comparing levels of contaminants in Ashland site media to more site-specific exposure pathways, toxicity reference values and receptors, rather than to conservative "toxicologically-based" (U.E. EPA 2001b) generic criteria. The assumptions used in the screening-level ecological risk assessment should be reviewed (e.g., 100% bioavailability) and modified as appropriate for site-specific and receptor-specific conditions (U.S. EPA 1997).

However, since there have been two ecological risks assessment already conducted (SEH 1998a; 2002), this step will be eliminated in this Baseline Problem Formulation. Chemicals that have been identified as contaminants in these earlier risk assessments will be the presumptive list for this Baseline Problem Formulation.

2.3 Other Factors of Potential Ecological Concern

In addition to contaminants that are related to past industrial operations on the site there are several other factors, resulting from various historical operations near the Ashland site, that cause conditions which may have some of the same effects as contaminants on habitat characteristics and thus on the receptors being evaluated in this BERA. While it may not be possible to quantify the effects of these other stressors relative to the effects from Ashland site contaminants it should be understood that the physical disturbances caused by these stressors may limit the degree of habitat recovery that would occur following any remediation. Thus, it is critical that the studies developed to collect data necessary to support this BERA consider that reference areas that are selected have the potential to help differentiate impacts caused by Ashland site contaminants from impacts due to other Ashland site stressors.

The following is a preliminary list of activities that have resulted in physical and chemical alterations to the habitat that are not directly related to MGP activities:

1) City-owned parcels of the lakefront were created during the late 1880s to the early 1900s by the uncontrolled placement of wood wastes, soil, sand, and demolition waste material into Chequamegon Bay;
2) Sawdust and wood waste from a series of sawmills that operated on the Ashland site from the early 1880s until about the mid-1930s were dispersed by natural forces, rain, flooding, storms and ice throughout Chequamegon Bay; and,
3) Log rafting and timber loading led to bark and wood waste accumulating to depths of many feet in various places in Chequamegon Bay.
4) Releases from wood treatment operations.
5) Discharges from Ashland POTW.

As reported by SEH (1998b), wood waste is found throughout the Ashland site in thickness averaging about 0.26 meters (~ eight inches) and ranging from 0-2 meters, "the wood chip layer extends from the Ashland Harbor shoreline out to the harbor mouth (approximately 270 m) and is deepest at the east end of the harbor (up to 2.1 m), rapidly declining to a thickness of between 15 and 24 cm [six-eight inches] over the majority of the remainder of the harbor."

In addition to the potential direct effects to habitat quality from these factors there are also indirect effects that may include release of excess nitrogen (which may exist in the form of ammonia, nitrate and/or nitrite) and phosphorous (in the form of phosphate). The presence of wood waste on top of the sediment bed can also increase biological oxygen demand and materially affect the dissolved oxygen levels at the sediment-water interface and at shallow depths within the sediment. Both the presence of excess nutrients, soluble organics and changes to dissolved oxygen can cause changes to the environment of both benthos and fish and may limit their presence in certain areas. Other physical changes related to changes in lake level, storms and sediment deposition dynamics also potentially can modify the characteristics of the aquatic environment and thus exert an effect on aquatic receptors.

The former Ashland POTW operated at the site of the former sawmill (at the end of Prentice Avenue), adjacent to the highest levels of sediment contaminants, until approximately 1990. It is likely that this operation resulted in varying compounds, including metals and SVOCs, being discharged into the Harbor area.

2.4 Risk Management Goal and Management Objectives

As defined by USEPA (2001), "a *risk management goal* is a general statement of the desired condition or direction of preference for the entity to be protected. It is often developed independently of the risk assessment process. [....], *management objectives*, while similar to management goals, differ in that they should be specific enough to use when developing assessment endpoints and measures."

Once broad risk management goals are developed, more specific risk management objectives are identified to define how the management goal is achieved and provide a basis for later risk management decisions (USEPA 2001).

Because multiple Interested Parties need to be involved in decision-making during the entire remedial investigation process, including the risk assessment and risk management, management goals should be developed as part of a process involving all Interested Parties. While the risk management goals need to recognize legislative requirements, additional factors such as economic considerations or public values may be integrated into these broad goals to recognize the uniqueness of each site.

The following risk management goal for the affected area at Ashland is proposed:

> **"Reduce to acceptable levels the risks to human health and the environment that may result from site-related contamination in the sediments at Ashland."**

The proposed management objectives that follow from this management goal are:

1) **Restore surface sediment quality so that it can support viable and self-sustaining populations of benthic macroinvertebrates and fish. This includes three corollary management objectives:**
    a. **Reduce levels of contaminants in surface sediment to a level that is compatible with supporting a diverse self-sustaining benthic macroinvertebrate community.**
    b. **Reduce or eliminate conditions resulting from the presence of wood waste overlying surface sediment that adversely affect the potential for benthic organisms to inhabit the sediment surface.**
    c. **Ensure contaminants in subsurface sediments are not transported to the sediment surface or water column, though diffusion, bioturbation, interstitial advection, erosion, resuspension or other transport mechanisms in quantities sufficient to jeopardize the sustainability of benthic macroinvertebrate and fish populations at Ashland and adjacent areas.**
2) **Ensure that the populations of bird and mammals that depend upon aquatic prey are not impacted from ingesting site-related chemicals in fish and invertebrates at**

Ashland to the extent that it will jeopardize the sustainability of these species' populations.

3) **Ensure protection of special status species by protecting individual representatives of these species from unacceptable acute and chronic exposures to site-related chemicals originating from Ashland sediments.**

These proposed management goals and objectives follow after a consideration of many potential management goals and objectives. Xcel Energy believes that these are the key management objectives for the Ashland site. While this list of management objectives could be more extensive, the achievement of these management objectives will ensure other biotic communities' environmental values are protected.

2.5 Assessment Endpoints

An assessment endpoint, according to USEPA (1997) is "an explicit expression of the environmental value [or ecological entity (USEPA 1998)] that is to be protected." Assessment endpoints should be closely related to the management objectives developed in Section 1.6. For specific assessment endpoints, risk hypotheses are evaluated using measures of exposure, effects, and ecosystem characteristics.

2.5.1 Assessment Endpoints for Aquatic Receptors

The following assessment endpoints are proposed for aquatic receptors:

1) Survival, growth, and reproduction of benthic macroinvertebrate communities in the affected areas at the Ashland site.
2) Survival, growth, and reproduction of fish communities in the affected areas of the Ashland site.

These two assessment endpoints are proposed for use in evaluating the attainment of the proposed management goal and the three management objectives. It is assumed that these assessment endpoints cannot be achieved without addressing the potential for adverse effects from both contaminants and wood waste in the surface sediment under present conditions as well as addressing the potential for contaminants in subsurface sediment to be remobilized and transported to the sediment surface or water column under some circumstances in the future. The approach to evaluate these assessment endpoints varies depending upon the management objective that is being addressed. This will be discussed in more detail in Step 4 of the ERAGs process: Study Design and Data Quality Objective Process.

2.5.2 Assessment Endpoints for Terrestrial Aquatic Prey-Dependent Receptors

Most of the management objectives relate directly to aquatic receptors, however if it is demonstrated that any of the COPCs biomagnify there is the potential for adverse effects to populations of aquatic prey-dependent wildlife, even though benthic macroinvertebrate and fish communities may not be impacted. Based upon the literature and site studies conducted to date this potential is remote, however, the following assessment endpoint will be proposed:

1) Survival, growth, and reproduction of aquatic-dependent wildlife in habitats bordering the Ashland site.

2.5.3 Assessment Endpoints for Special Status Species

The following assessment endpoint is proposed for special status species:

1) Survival and growth of individuals of special status species.

2.6 Conceptual Site Model

In accordance with EPA guidance (USEPA 1997; 1998), the conceptual site model (CSM) is intended to describe key relationships between stressors and assessment endpoints, including the sources and releases of contaminants, the fate and transport of contaminants, and the exposure pathways that link contaminants to ecological receptors. It is typically both a graphical representation and narrative discussion of primary exposure pathways of contaminants to ecological receptors. The CSM provides a framework for predicting effects on ecological receptors and a basis for generating risk questions and testable hypotheses. The CSM also provides a basis for identifying data gaps and designing the necessary studies to fill them.

2.6.1 Contaminant Fate and Transport

Information on how contaminants are potentially transported or transformed in the environment physically, chemically and biologically, is used to identify potential exposure pathways to ecological receptors.

2.6.1.1 Conceptual Model of Sediment Dynamics

Contamination in Ashland Harbor sediments appears to be mainly confined to a sediment layer extending a few hundred feet from the shoreline. The layer has a maximum thickness near the shoreline, typically 3 to 4 feet, and tapers off in the offshore direction. The layer is characterized by the presence of wood chips and most of the contamination is confined to this layer, although some contamination appears to exist just below the wood chip layer. The wood chips apparently were derived from local material, either from the sawmill or directly from logs floated into and rafted in the Ashland Harbor area.

Based upon a review of existing data, there are at least two possible conceptual models that explain the current contaminant distribution at the Ashland site contamination. These models are preliminary, but serve as a starting point for the empirical analysis.

In the first conceptual model, it is noted that much of the existing shoreline and some of the marina structures were created by back-filling soil into the bay. It is possible that the process of back-filling, which is assumed to have occurred episodically between 120 and 60 years ago, created most of the contaminated wood-chip laden sediment layer. The backfill material, which was likely generated from the bluff and surrounding area, contained wood processing wastes as well as contamination from facilities operating in the area. Xcel Energy has produced documentation indicating that the PAH contamination measured in the sediments has been generated from other sources (e.g., wood treatment at the former Schroeder Lumber Company) in addition to the former manufactured gas plant. As this material was transported to the harbor, some of it escaped into the surface water and settled out in the near shore area. The shape of the wood-chip layer, thick near the shoreline and tapering offshore, is consistent with this view.

In this interpretation, most of the contamination was derived from existing soil and surface contamination associated with the back-fill material. It is also possible that sediments associated with surface runoff and groundwater transport contributed to the development of the deposit. Both contaminated and uncontaminated sediments reached the Ashland site from rainfall induced surface runoff originating in the watershed adjacent to the Ashland site. It is alternatively possible that contaminated surface runoff mixed with re-suspended sediments and contributed to the contaminated sediment layer as it evolved. However, it is likely that these processes only played a secondary role relative to the contamination derived from the back-filling.

If this interpretation is substantiated, it may imply that the sediments are fairly stable. The contaminated layer would not require historic or ongoing active sediment transport to have developed, and therefore it is possible the transport is low and insignificant in the area and sediments are relatively stable.

An alternate conceptual model for the evolution of the contaminated sediment layer is based on regional sediment transport patterns. In this interpretation, much of the sediment that comprises the layer may have originated up and down shore of the Ashland site. Sediment was (and possibly still is) transported via waves and currents all along the shoreline and during high energy events and the sediment that makes its way into the Ashland site will deposit during the waning phase of the event. Although the sediment may have been contaminant-free at its origin, it likely mixed with contaminated runoff and/or contaminated sediments from the watershed adjacent to the Ashland site, and then deposited at the Ashland site.

It is known that logging was active during the last 120 years in the region. Logging could have provided a steady source of sediment to the harbor area since logging activities are known to increase soil erosion and provide additional source of sediments to rivers. A review of stream and river networks in the area show two drainage basins, one to the east and a larger one to the west. These rivers may have carried relatively large sediment loads to Chequamegon Bay, some of which eventually were deposited at the Ashland site. It is likely that the current sediment load has been reduced relative to loads that occurred during the period of relatively uncontrolled historic logging activities, due either to reduced logging activities and/or improvement in logging procedures.

In this conceptual model, the evolution of the contaminated sediment layer occurred fairly continuously, due primarily to the sediment loads associated with the regional logging industry. In a large-scale long-term view, for the period of 120 to 60 years ago, the Bay was unable to flush the anthropogenic source of sediments at the rate that they were supplied. The hydrodynamic forces may have been able to remove some of the additional load to deeper water but not all of it. Thus the sediments began to accumulate along the shoreline as well as in deeper waters in the Bay. In terms of sediment balance, there was net sediment input into the Bay. If the logging industry based sediment load has actually been reduced, then the sediment balance near the coastline may begin shifting from depositional to erosional. However, the erosion rates may be relatively low, due to the relatively low hydrodynamic energy and limited fetch lengths in the area. In either case, quantification of this conceptual model will determine net erosion or depositional rates.

2.6.1.2 Environmental Fate of Contaminants of Concern

<u>*Polychlorinated Aromatic Hydrocarbons*</u>

<u>(Write-up on this group of COPCs provided as an example)</u>

Polycyclic aromatic hydrocarbons are a diverse class of organic compounds that include about one hundred individual substances containing two or more fused benzene, or aromatic, rings. Low molecular weight (LMW) PAHs are PAHs with fewer than four rings, while high molecular weight (HMW) PAHs have four or more rings. The LMW PAHs include acenaphthene, acenaphthylene, anthracene, fluorene, naphthalene, 2-methylnaphthalene, and phenanthrene. The HMW PAHs include benz(a)anthracene, benzo(a)pyrene, chrysene, dibenz(a,h)anthracene, fluoranthene, and pyrene.

The behavior of PAHs in surface waters depends on a number of environmental factors including both chemical-specific and site-specific factors. The physico-chemical properties of PAHs tend to determine their fate in aquatic systems. The PAHs with high solubilities largely remain dissolved in surface water. PAHs with lower solubilities usually become associated with suspended particulates and eventually become incorporated in the bed sediment.

While in the water column either in association with colloidal material or suspended particulates, their fate tends to be governed by physical hydrodynamic factors, i.e. advective transport. While in the water column PAHs potentially may be transported to other areas, biodegrade, evaporate, photochemically degrade or may be incorporated into water column biota.

Sediments are the major environmental sink for PAHs. Deposition of PAHs associated with suspended particulates occurs relatively rapidly and they are incorporated into the bed sediment. Release of some materials such as coal tar or creosote, which contain a number of PAH compounds, into an aquatic environment can lead to relatively quick incorporation. Once in the sediment bed release of lower weight PAHs into the overlying water column is possible although the primary fate is biodegradation and biotransformation by benthic organisms (USEPA 1980 as cited by Eisler 2000). The rate of these biodegradation processes vary substantially however, depending upon the molecular weight of the PAHs and the presence of microbial communities in the sediments. In the absence of penetration radiation and oxygen (beneath the immediate surface layers) degradation rates are typically slow. As a result at most historical PAH waste sites, PAHs are found distributed relatively deeply in the sediment column since the rate of sediment deposition greatly exceeds PAH degradation rates. This suggests two factors that need to be considered in the CSM for the site: 1) Are PAHs being buried sufficiently rapidly that they are no longer bioavailable to benthos and fish that are only in contact with the surficial sediments; and, 2) Is the sediment bed sufficiently stable that once incorporated into the sediment bed, the PAHs are substantially sequestered from future exposure to aquatic receptors.

Releases of PAHs into aquatic ecosystems pose a number of potential risks to aquatic organisms. At sufficiently elevated levels waterborne PAHs can be lethal to water column receptors and long-term exposure to sublethal levels have been shown to affect survival, growth and reproduction. Exposure to PAHs in the sediment can adversely affect the survival, growth, and reproduction of benthic invertebrates and also have been correlated with the presence of an increased incidence of neoplasms and histopathological effects in wild fish. While in most studies of these latter effects the presence of significant amounts of other contaminants precluded establishing a direct causal relationship between the PAHs in the sediments and these observed effects, there is some weight of evidence that relatively high levels PAHs can have these effects on fish, particularly demersal fish.

With regard to the potential for wild fish to bioaccumulate PAHs, Eisler (2000), citing two primary sources, indicates that, "PAH levels in fish are usually low because this group rapidly metabolizes PAHs (Lawrence and Weber 1984a); furthermore, higher molecular weight PAHs ... do not seem to accumulate in fish (West et al. 1984)."

Findings from numerous laboratory and field studies have also indicated that 3-5 ring PAHs such as phenanthrene and benzo[a]pyrene are extensively metabolized in fish tissues. Within 24 hours, metabolism can convert up to 99% of the parent PAH to metabolites, which are subsequently excreted into bile, a major route of elimination (Meador et al. 1995). Although a number of environmental factors influence the rates of uptake, metabolism, and excretion of PAHs in fish (Varanasi et al. 1989; Meador et al. 1995), the general lack of bioaccumulation of PAHs limits exposure to higher trophic level organisms via the food chain (e.g., Eisler 2000; McElroy et al., 1989; Suedel et al. 1994).

As reported in a recent paper by Burkhard and Lukasewycz of USEPA's National Health and Environmental Effects Research Laboratory (Burkhard and Lukasewycz 2000), "an extensive but unsuccessful literature search was performed for field-measured bioaccumulation factors (BAFs) and biota-sediment accumulation factors (BSAFs) for polycyclic hydrocarbons (PAHs); no reported values were found for fish. The lack of BAFs and BSAFs for PAHs occurs in part because PAHs are metabolized by fish, resulting in very low or non detectable concentrations of the parent PAHs in fish tissues [Varanasi et al. 1989]." Typical BSAFs that are available include: phenanthrene (0.00011), fluoranthene (0.00016), pyrene (0.0071, benzo(a)pyrene (0.0054), and chrysene/triphenylene (0.00033) (Burkhard and Lukasewycz 2000).[2]

[2] While there has been concern expressed in recent studies that the metabolites of PAHs may have effects on fish and, to some extent, to upper levels of the food chain, it is assumed that the TRVs for sediments proposed above will

Because PAHs and other Type I narcotic chemicals[3] are largely metabolized in prey species, i.e. fish, trophic transfer of these chemicals further up the food chain is not a likely factor in driving risk management decisions. The one study that is frequently used to develop a TRV for PAHs for birds is a study by Patton and Dieter (1980) (as reported in Eisler (1987). Quoting Eisler (1987): "[i]n one study, Patton and Dieter (1980) fed mallards diets that contained 4,000 mg PAHs/kg (mostly as naphthalenes, naphthenes, and phenanthrene) for a period of 7 months. No mortality or visible signs of toxicity were evident during exposure; however, liver weight increased 25% and blood flow to liver increased 30%, when compared to controls. This suggests a low likelihood of effects to piscivorous birds."

Indeed, the doses that may lead to lethal and sublethal effects in birds and mammals tend to be much higher than those that cause neoplasms. (i.e., up to an order or magnitude higher; ATSDR 1990). In mice, ingestion of diets containing 50 to 250 mg/kg benzo[a]pyrene for 70 to 197 days resulted in a > 70% incidence of stomach tumors (ATSDR 1990).

Some aquatic receptors do not readily metabolize PAHs however; bivalves are one group of invertebrates that have demonstrated some capability of accumulating PAHs (Eisler 1987; 2000).

2.6.2 Ecosystems Potentially at Risk

This BERA focuses on a portion of the aquatic ecosystem of Chequamegon Bay, Lake Superior in the vicinity of Ashland Wisconsin (Figure 2). The boundaries of this "affected habitat" have not yet been well established. It has not been determined whether contaminants associated with the sediments of this portion of the Ashland site are transported in significant quantities beyond that immediate area either through sediment erosion and advective transport or through food chain transfer. Studies recommended in this BERA will help answer this question.

2.6.3 Exposure Pathways

Exposure pathways are routes by which contaminants are transferred from a contaminated medium to ecological receptors. For the Sediment Operable Unit of Ashland Lakefront Site, the potential routes by which ecological receptors may be exposed to Ashland site contaminants are illustrated in the CSM (Figure 3). The CSM illustrates those exposure pathways that are potentially complete and will therefore be evaluated in this BERA. These include the following:

- Birds - ingestion of sediment, surface water, and food;
- Mammals- ingestion of sediment, surface water, and food;
- Fish - ingestion and direct contact with sediment and surface water;

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be protective of effects from PAH metabolites as well. While acknowledging that metabolism of some PAH molecules may increase the relative toxicity of a particular compound, the metabolism of PAHs generally increases the potential for excretion or elimination from an organism and it is assumed that this enhanced potential for elimination offsets, to some extent, the toxicity of PAH metabolites.

[3] PAHs are not narcotics in <u>all</u> aquatic species; they have specific modes of action in many cases.

- Reptiles and amphibians - ingestion and direct contact with sediment and surface water and ingestion of food;
- Aquatic invertebrates - ingestion and direct contact with sediment or surface water and ingestion of food;
- Aquatic plants - root uptake and direct contact with sediment and surface water; and,
- Phytoplankton and zooplankton – direct contact with surface water.

Pathways deemed to be most important are shown as bolder lines in CSM (Figure 3). Some exposure pathways have been combined with others or cannot be quantitatively evaluated because of a lack of available information for the exposure evaluation. These will be considered uncertainties in this BERA. Examples of these potential exposure pathways include dermal and inhalation exposures for birds and mammals. Although these pathways are not quantitatively evaluated they are considered relatively minor exposure pathways relative to other exposure pathways.

Aquatic invertebrates, including benthic, epibenthic, pelagic and planktonic invertebrates, may be exposed to chemicals in sediment and surface water through ingestion and direct contact or by absorption through their skin. They can also be exposed through their food. Aquatic plants potentially can absorb chemicals from sediment and surface water through their roots, leaves, or stems. Both aquatic invertebrates and aquatic plants can serve as a major exposure pathway to upper trophic levels since they are prey for fish, birds, and mammals; this is termed trophic (or food chain) transfer. Food chain transfer of chemicals is important only for those chemicals that are bioaccumulative.

Amphibians and reptiles may be exposed to chemicals in sediment and surface water along the shoreline through ingestion, dermal contact, and by feeding on contaminated aquatic invertebrates. Exposure may occur during feeding, early development of eggs and larvae, or burrowing. Amphibians and reptiles also may be an exposure pathway to birds and mammals through food chain transfer.

Fish may be exposed to chemicals in sediment and surface water through ingestion, dermal contact, uptake through gills, and by feeding on aquatic plants, invertebrates, or smaller fish. Exposure may occur during feeding, spawning, or burrowing. Aquatic vertebrates also may be an exposure pathway to birds and mammals through food chain transfer.

Birds and mammals may be exposed directly to chemicals in the sediment and surface water through incidental ingestion, dermal contact, and inhalation of particulates, although the latter exposure pathway will not be quantitatively evaluated. They may also be exposed indirectly through food chain transfer although as discussed previously, this exposure pathway is significant only for those chemicals that are bioaccumulative.

The following exposure pathways will not be quantitatively evaluated for the following reasons (Table 1):

Table 1. Exposure Pathway Evaluation.

Potential Exposure Pathway	Reason for not Evaluating Quantitatively
Microbial processes: Exposure to chemicals in sediment and surface water.	Inadequate information to quantitatively evaluate.
Benthic and aquatic invertebrates: Exposure to chemicals through food chain transfer.	Inadequate information to quantitatively evaluate. Fish tissue will integrate any food chain transfer at lower trophic levels.
Birds and Mammals: Exposure to chemicals through dermal adsorption.	The fur-covered skin of mammals and the feathers of birds limit the direct dermal uptake of chemicals from the environment and this pathway will not be evaluated. Preening and grooming behaviors, however, contribute to the incidental ingestion of soil or sediment, and are included as part of the incidental ingestion exposure pathway.
Birds and Mammals: Exposure to chemicals through inhalation.	It is doubtful that there is sufficient volatilization of sediment-associated chemicals to result in a threat to bird and mammal receptors. This will be discussed as an uncertainty.

2.6.4 Receptors of Concern (ROCs)

As part of the Baseline Problem Formulation receptors at risk within the affected habitat are identified from the conceptual site model. From these species several representative species that meet the following criteria are selected as receptors of concern (ROCs). These ROCs will be used in the BERA to evaluate the potential for adverse effects and serve as a proxy for other receptors that have similar niches; food habits or feeding behaviors, they are exposed to Ashland site contaminants in a similar manner or have similar sensitivity to Ashland site contaminants.

USEPA (1997; 1998) provides guidance on selecting these ROCs and indicates they should include:

- Resident species or communities exposed to the highest chemical concentrations in sediments and surface water;
- Species or functional groups that are essential to, or can be used as an indicator of the normal functioning of the affected habitat; and,
- Species of special concern, e.g. federal or state threatened or endangered species.

In the affected area of the Ashland site the receptors at risk include microbial communities, benthic macroinvertebrate communities and potentially fish. In addition, wildlife, primarily fish and invertebrate-eating waterfowl, may also be at risk. The various groups that meet the above selection criteria and from which ROCs are chosen are summarized below:

2.6.4.1 Microbial Communities

Microbial communities consist of bacteria, protozoans, and fungi. These microorganisms are intimately associated with the sediment and play a number of key roles in the cycling and transformation of nutrients in sediments and the water column. In addition, these communities degrade and transform detrital matter, support primary productivity, and mediate the sulfur cycle in the aquatic environments.

This functional group will be considered collectively as a ROC but will not be evaluated quantitatively.

2.6.4.2 Benthic Macroinvertebrate Communities

Benthic macroinvertebrates live on and in the sediment and are, thus, intimately associated with the sediment. They represent key elements of the aquatic food web because they consume aquatic plants and detritus and serve themselves as the food base for fish and to some extent birds and mammals.

This functional group will be considered collectively as a ROC. In addition, the fingernail clam, *Pisidium* sp., the most abundant mollusk found at the site (SEH 1998b), will be considered a ROC because of the potential for mollusks to bioaccumulate PAHs.

2.6.4.3 Fish Community

Fish represent an important element of the aquatic food web, processing energy from plants and detritus to fish-eating wildlife. Based upon a study of the fish community of the Ashland site by the U.S. Geological Survey Biological Resources Division from 1973 to 1996 a number of fish species occupying a variety of ecological niches were reported to utilize Chequamegon Bay, one community apparently occupying shallow water (< 3.0 m) and another fish community inhabiting deeper water (> 3.0m) (SEH 1998b). Species (e.g., black bullhead and long nose sucker) that feed on the benthos living in the affected area could contribute to food chain transfer of contaminants to upper trophic levels. In addition some species that typically spawn in shallow waters could potentially utilize Chequamegon Bay.

Two species of fish were identified as ROCs. The black bullhead (*Ictalurus melas*), a bottom feeding species, and the walleye (*Stizostedion vitreum*), an upper trophic level species that could potentially utilize the shallow waters of the Ashland area for foraging or to spawn.

2.6.4.4 Aquatic Dependent Wildlife: Birds and Mammals

It is unlikely that birds and mammals are exposed directly to contaminants in the sediment and surface water because the riparian areas along the shore are for the most part consist of engineered materials, i.e., rip rap or bulkheads, and do not provide foraging habitat. In addition, surface water concentrations of contaminants are less than water quality criteria (SEH 1998a: Table 6). However, some species, including wading birds, waterfowl, and some mammals may be potentially exposed to contaminants through food chain transfer by eating aquatic prey. Many species forage in aquatic or riparian habitat and consume a variety of aquatic invertebrate as well as fish species.

Two species of aquatic-dependent birds will be selected as ROCs, one, the great blue heron (*Ardea herodias*), a wading bird that feeds primarily on fish and amphibians, the other, a mallard duck (*Anas platyrhynchos)*, which feeds on benthic invertebrates and vegetation.

One species of an aquatic-dependent mammal that will also be selected as a ROC includes the mink (*Mustela vison)*. Mink are primarily piscivorous and while the shoreline area doesn't have the riparian cover habitat that mink prefer, they will be used as a ROC because it is expected that an exposure scenario using mink would provide a conservative evaluation of potential exposure to other aquatic-dependent mammals.

2.6.4.5 Special Status Species and Critical Habitats

A preliminary evaluation of Wisconsin and federal listed threatened, endangered or rare species with special status that may potentially be found in the Ashland Site area was conducted. Of those listed for Ashland County only the merlin, *Falco columbarius* would potentially be found in the Ashland site area. The merlin was therefore selected as a ROC.

2.6.4.6 Other Species not selected as ROCs

The following groups of receptors will not be considered as ROCs in the BERA (Table 2).

Table 2. Species Potentially at Risk but not Proposed as Receptors of Concern.

Functional Group	Reason For Not Including As Primary ROC
Aquatic Macrophytes	1) Much of the shoreline is engineered protection and the remaining portions of the harbor are too deep to support a community of macrophytes. 2) Protection of microbial and benthic macroinvertebrate communities and ROCs will result in protection of potential aquatic macrophytes and macrophyte habitat

Functional Group	Reason For Not Including As Primary ROC
Phytoplankton & Zooplankton	1) Populations of plankton are transitory, moving rapidly through the area with currents. It is extremely unlikely that even acute effects in this immediate area would have a material effect on plankton populations over broader areas since they have high reproductive capacities and turnover rates. 2) Levels of contaminants in surface water, the exposure medium for plankton, are within ambient water quality guidelines
Periphyton	Protection of microbial communities, benthic macroinvertebrate communities and ROCs will result in protection of periphyton.
Amphibians and reptiles	The riparian habitat is such that it is unlikely to support substantial numbers of reptiles and amphibian and they are unlikely to play a significant role in a food web that is based upon fish or macroinvertebrates from the affected area.

2.7 Risk Questions and Risk Hypotheses

Risk questions are questions about the relationship between assessment endpoints and their predicted responses when exposed to contaminants (USEPA 1997). Risk questions are important as they provide a more easily understood description of the basis for risk management decisions. However, since risk questions don't provide quantitative decision criteria for answering them, they are typically re-stated as risk hypotheses. By stating them as risk hypotheses it is possible to develop action levels and decision rules that make explicit what error rates are acceptable for accepting or rejecting these hypotheses. This will be discussed in more detail Step 4 of the ERAGS process: "Study Design and Data Quality Objective Process". By establishing objective criteria for accepting and rejecting risk hypothesis the temptation to exploit this uncertainty when making risk-management decisions is eliminated.

Key risk questions and risk hypotheses proposed for the Ashland site are summarized in Table 3.

Table 3. Assessment Endpoints, Risk Questions and Testable Hypotheses.

Assessment Endpoint	Risk Questions	Testable Hypotheses
Survival, growth, and reproduction of benthic macroinvertebrate communities.	Are concentrations of contaminants in the sediments and pore water at the Ashland site sufficiently elevated that they cause adverse effects on benthic macroinvertebrate survival, growth and reproduction?	Sediment contaminant concentrations are elevated ($p<0.05$) above the no-observable-effects-concentrations (NOEC) or lowest-observable-effects concentrations (LOEC) for benthic biota.
		Sediment pore water contaminant concentrations are elevated above ($p<0.05$) the water-only NOECs and LOECs for benthic species.
		Sediments of the affected area of the Ashland site have elevated ($p<0.05$) toxicity to surrogates for resident macroinvertebrate species compared to sediments in reference areas.
		Benthic communities inhabiting sediments at the Ashland Site are impaired when compared to benthic communities inhabiting reference area sediment.
Survival, growth, and reproduction of fish communities.	Are concentrations of contaminants in the sediments at the Ashland site sufficiently elevated that they cause adverse effects to fish survival and growth?	Sediment concentrations are elevated ($p<0.05$) above NOECs or LOECs for fish.
		Sediments at the Ashland site have elevated ($p<0.05$) toxicity to surrogates (fathead minnow) for fish species compared to sediments in reference areas.
		Sediment contaminant concentrations are greater than ($p<0.05$) levels that result in deformities, fin erosion, or other histopathological effects in fish.
		Tissue residues of contaminants are greater ($p<0.05$) in fish utilizing sediments at the Ashland site than in fish from reference areas.

Assessment Endpoint	Risk Questions	Testable Hypotheses
		Tissue residues of contaminants are greater ($p<0.05$) in fish utilizing sediments at the Ashland site than critical body residue effects levels as derived from the literature.
Survival, growth, and reproduction of aquatic-dependent wildlife.	Are concentrations of contaminants in the sediments incidentally ingested and the diet of aquatic-dependent wildlife sufficiently elevated that they cause adverse effects to their populations?	Intake of contaminants ingested with prey and incidental sediment is greater ($p<0.05$) than the no effect dose (NOAEL) or low effect dose (LOAEL) to aquatic-dependent wildlife.
Survival, growth and reproduction of individuals of special status species.	Are concentrations of contaminants in the sediments incidentally ingested and diet of aquatic-dependent wildlife sufficiently elevated that they cause adverse effects to individuals of special status species?	Intake of contaminants in prey and incidental sediment is greater ($p<0.05$) the NOAEL or LOAEL for aquatic-dependent wildlife.

3.0 STEP 4: STUDY DESIGN AND DATA QUALITY OBJECTIVE PROCESS

As indicated in USEPA (1997), "Step 4 of the ecological risk assessment establishes the measurement endpoints [...], completing the conceptual model begun [earlier]. Step 4 also establishes the study design [...] and data quality objectives based upon statistical considerations." In this section of the Straw Man Baseline Problem Formulation, Xcel Energy will discuss the measurement endpoints proposed to evaluate the assessment endpoints from Step 3 and propose the study design and data quality objectives to evaluate the proposed measurement endpoints. Xcel Energy believes that the studies proposed here are adequate to provide the necessary information to address the risk management goal and support risk management decision-making.

3.1 Lines of Evidence and Measurement Endpoints

A line of evidence is, "information derived from different sources or by different techniques that can be used to describe and interpret risk estimates" (USEPA 1998). USEPA (1997) concluded that, in general, there are four possible lines of evidence that can be used to test these risk questions and hypotheses:

Comparing estimated or measured exposure levels of contaminants with levels of the contaminants that are known to cause adverse effects to receptors associated with the assessment endpoints;
Comparing laboratory bioassays on media from the subject site with artificial media or media from a reference site;
Comparing in situ toxicity tests at the subject site with in situ toxicity tests at a reference site; and,
Comparing observed effects in receptors associated with the subject site with similar receptors at a reference site. This could include population and community studies, for instance.

Each of these lines of evidence can incorporate one or more measurement endpoints that describe the change in the assessment endpoint in response to exposure to a stressor, in this case, a COPC. As defined in USEPA (1997), a measurement endpoint is, "a measurable ecological characteristic that is related to the valued characteristic chosen as the assessment endpoint." The lines of evidence and measurement endpoints proposed for the Ashland site are discussed in more detail in the following sections. The lines of evidence proposed for use in this baseline ecological risk assessment include:

Exposure Levels: Concentration Values In Environmental Media

This approach is essentially a predictive line of evidence. By comparing levels of contaminants measured in site media, e.g., sediment and surface water, or organisms, e.g., contaminant levels in fish and invertebrate tissue, to toxicological information from the literature, one can predict the likely response of site-specific receptors. Uncertainty associated with this approach relates to the differences between the site-specific exposure and the conditions of the study from which the toxicological information was obtained.

Site-Specific Toxicity Data: Laboratory Bioassays

This line of evidence generally uses tests of receptor response to site media, e.g. a sediment, soil or water column toxicity bioassay. This approach is often more relevant than the literature in understanding the responses of site receptors to contaminants. However, since site-specific exposure conditions cannot be exactly replicated, nor do test organisms necessarily react to a stressor similar to site receptors, there is some uncertainty associated with this approach.

Direct Observations of Receptor Populations

Direct observations on presence, composition, condition and behavior of receptors populations and communities at the site of interest can also be a fairly direct line of evidence for evaluating the extent of site-related impacts on ecological receptors. The inherent variability in natural populations, however, inevitably introduces substantial uncertainty into interpretation of this line of evidence and it is critical that analysis of this line of evidence be sufficiently sensitive to distinguish differences between site and reference populations or communities that may be related to the stressors being evaluated from differences in these populations or communities that are only due to natural variability.

3.1.1 Proposed Measurement Endpoints

The measurement endpoints proposed for the affected areas of the Ashland site are summarized in Table 4.

Table 4. Risk Questions, Testable Hypotheses and Measurement Endpoints.

Assessment Endpoint	Risk Questions	Testable Hypotheses	Measurement Endpoint
Survival, growth, and reproduction of benthic macroinvertebrate communities.	Are concentrations of contaminants in the sediments and pore water at the Ashland site sufficiently elevated that they cause adverse effects on benthic macroinvertebrate survival, growth and reproduction.	Sediment contaminant concentrations are elevated ($p<0.05$) above the no-observable-effects-concentrations (NOEC) or lowest-observable-effects concentrations (LOEC) for benthic biota.	Comparison of contaminant concentrations in sediment to toxicity reference values for sediment.
		Sediment pore water contaminant concentrations elevated above ($p<0.05$) the water-only NOECs and LOECs for benthic species.	Comparison of contaminant concentrations in sediment pore water (0.45 um filtered) to toxicity reference values for surface water.
		Sediments at the Ashland site have elevated ($p<0.05$) toxicity to surrogates for resident macroinvertebrate species compared to sediments in reference areas.	*Hyalella azteca* and *Chironomus tentans* response during exposure to contaminated sediment.
		Benthic communities inhabiting sediments of the affected area at the Ashland site are impaired compared to benthic communities inhabiting reference area sediment.	Abundance, distribution and species composition of benthic macroinvertebrate communities relative to communities inhabiting reference area sediments.
Survival, growth, and reproduction of fish communities.	Are concentrations of contaminants at the	Sediment concentrations are elevated ($p<0.05$) above NOECs or LOECs for fish.	Comparison of contaminant concentrations in sediment to toxicity reference values for sediment.

Assessment Endpoint	Risk Questions	Testable Hypotheses	Measurement Endpoint
	Ashland site sufficiently elevated that they cause adverse effects to fish survival and growth?	Sediments at the Ashland site have elevated ($p<0.05$) toxicity to surrogates (fathead minnow) for fish species compared to sediments in reference areas.	Fathead minnow response during exposure to contaminated sediment.
		Sediment contaminant concentrations are greater than ($p<0.05$) levels that result in deformities, fin erosion, or other histopathological effects in fish.	Comparison of contaminant concentrations in sediment to levels that are associated with deformities, fin erosion, or other histopathological effects in fish as derived from the literature. OR Evaluation of fish from the Ashland site area for deformities, fin erosion, or other histopathological effects.
		Tissue residues of contaminants are greater ($p<0.05$) in fish utilizing sediments at the Ashland site than in fish from reference areas.	Comparison of fish tissue residues to tissue residues of fish inhabiting reference area sediment.
		Tissue residues of contaminants are greater ($p<0.05$) in fish utilizing sediments at the Ashland site than critical body residue effects levels as derived from the literature.	Comparison of fish tissue residue levels to critical body residue toxicity reference values.

Assessment Endpoint	Risk Questions	Testable Hypotheses	Measurement Endpoint
Survival, growth, and reproduction of aquatic-dependent wildlife.	Are concentrations of contaminants in the sediments and diet of aquatic-dependent wildlife sufficiently elevated that they cause adverse effects to their populations?	Intake of contaminants ingested with prey and incidental ingestion of sediment is greater ($p<0.05$) than the no effect dose (NOAEL) or low effect dose (LOAEL) to aquatic-dependent wildlife.	Comparison of exposure concentrations to toxicity reference values.
Survival, growth and reproduction of individuals of special status species.	Are concentrations of contaminants in the sediments and diet of aquatic-dependent wildlife sufficiently elevated that they cause adverse effects to individuals of special status species?	Intake of contaminants in prey and incidental ingestion of sediment is greater ($p<0.05$) than the NOAEL or LOAEL for aquatic-dependent wildlife.	Comparison of exposure concentrations to toxicity reference values.

3.2 Data Gap Analysis

The gap analysis and the study design is the culmination of the Problem Formulation and Study Design stages of the BERA. Once the management goals, management objectives, and their associated assessment and measurement endpoints have been agreed to as part of a process involving all Interested Parties, then the risk team can establish what data need to be collected to support the risk assessment and ultimately form the basis for risk management decision making. Following the gap analysis an action plan can be developed that provides the details of sampling design, analytical protocols and quality control objectives can be prepared. These are eventually incorporated into a Work Plan and a Sampling and Analysis Plan. Table 5 identifies data gaps based upon the risk assessment endpoints selected.

Table 5. Data Gaps Analysis for Risk Assessment Endpoints.

Management Objective	Assessment Endpoint	Measurement Endpoint	Information Needed
Restore surface sediment quality so that it can support viable and self-sustaining populations of benthic macroinvertebrates and fish. AND Ensure contaminants in subsurface sediments are not transported to the sediment surface or water column, though diffusion, bioturbation, interstitial advection, erosion, resuspension or other transport mechanisms in quantities sufficient to jeopardize the sustainability of benthic macroinvertebrate and fish populations. AND Reduce or eliminate conditions resulting from the presence of wood waste overlying surface sediment	Survival, growth, and reproduction of benthic macroinvertebrate and fish communities.	Comparison of contaminant concentrations in sediment to toxicity reference values for sediment.	Surface sediment (0-10cm) concentrations for all contaminants. Additional data are needed in order to adequately estimate spatial parameters for statistical comparison between Ashland sediments as well as to develop a reliable characterization of contaminant spatial gradients in the affected area. An evaluation of sediment stability is proposed to evaluate the potential for contaminants at depth in the affected area of the Ashland site to be remobilized by Ashland site dynamics. This needs to consider the heterogeneity of the site and the fact that sediment stability is not expected to be uniform across the site. In addition, it is necessary to understand whether the presence of contaminants or wood waste is responsible for failure to achieve management objectives. This requires careful selection of the reference area.
		Comparison of contaminant concentrations in sediment pore water (0.45 um filtered) to toxicity reference values for surface water.	Porewater concentrations (0-10cm) for all contaminants. No pore water currently exists. Representative pore water data is needed to corroborate estimates of contaminant bioavailability.

Management Objective	Assessment Endpoint	Measurement Endpoint	Information Needed
that adversely affect the potential for benthic organisms to inhabit the sediment surface.		*Hyalella azteca, Chironomus tentans* and fathead minnow response during exposure to contaminated sediment.	Sediment bioassays have already been conducted and may be adequate.
		Abundance, distribution and species composition of benthic macroinvertebrate communities relative to communities inhabiting reference area sediments.	Past studies of benthic community structure have not been sufficiently statistically rigorous. If this is to be used as a material "line of evidence" then study design must be re-evaluated after decision rules are established to define acceptable level of error. Otherwise use of this line of evidence is limited.
		Comparison of contaminant concentrations in sediment to levels that are associated with deformities, fin erosion, or other histopathological effects in fish as derived from the literature. OR Evaluation of fish from the Ashland site area for deformities, fin erosion, or other histopathological effects.	Develop information from literature on concentrations of PAHs associated with deformities, fin erosion, or other histopathological effects in fish. AND/OR Data needed on deformities, fin erosion, or other histopathological effects in fish from the Ashland site.

Management Objective	Assessment Endpoint	Measurement Endpoint	Information Needed
		Comparison of tissue residue levels of contaminants in fish inhabiting Ashland site sediment to critical body residue effects levels as derived from the literature and to tissue residue levels in fish from reference areas.	Results from fish collected to support original ecological risk assessment (SEH 1998a) should be adequate.
Ensure that the populations of bird and mammals that depend upon aquatic prey are not impacted from ingesting site-related chemicals in fish and invertebrates at Ashland to the extent that it will jeopardize the sustainability of these species' populations.	Survival, growth, and reproduction aquatic-dependent wildlife.	Comparison of exposure concentrations to toxicity reference values.	Information on feeding ecology and area use for aquatic-dependent wildlife relating to home range, foraging range, and intake rates are available in the literature Estimates of tissue residues in representative prey species. This is available for fish but not for benthic invertebrates or plants which would serve as potential food for mallard ducks.

Management Objective	Assessment Endpoint	Measurement Endpoint	Information Needed
Ensure protection of special status species by protecting individual representatives of these species from unacceptable acute and chronic exposures to site-related chemicals originating from Ashland sediments.	Survival and growth of individuals of special status species.	Comparison of exposure concentrations to toxicity reference values.	Information on feeding ecology and area use for aquatic-dependent wildlife relating to home range, foraging range, and intake rates are available in the literature Estimates of tissue residues in representative prey species. This is available for fish but not for benthic invertebrates or plants which may serve as potential food for special status species.

3.3 Study Design

Based upon the above analysis additional studies are proposed fill data gaps. Concepts for the proposed studies are provided in the following sections.

3.3.1 Surface Sediment Characterization

Additional data relating to the presence and distribution of contaminants in surface sediment are needed for a reliable statistical comparison between Ashland surface sediments and reference areas, and to characterize contaminant spatial gradients within the affected area of the Ashland. This characterization should include other sediment parameters that may affect bioavailability of contaminants, e.g., organic carbon, acid volatile sulfides, etc.

3.3.2 Pore Water Characterization

The primary exposure medium for benthic organisms is pore water. Although bioavailability of contaminants in sediment can be indirectly estimated using an equilibrium partitioning approach, measurement of contaminants in pore water, at least in representative locations, is a more accurate method for characterizing the exposure medium for these organisms. To date there has been no evaluation of pore water in the affected area of the Ashland site. It is proposed that this be considered and incorporated into the supplemental sampling plan.

3.3.3 Sediment Bioassays

Xcel Energy believes that the sediment bioassays conducted in 2001 as part of the ecological risk supplemental studies (SEH 2002) are adequate as a site-specific toxicity line of evidence.

3.3.4 Benthic Community Characterization

This design of this study has to consider both the potential impacts of contaminants in surface sediment as well as those of the wood waste located on and throughout the sediment bed at the Ashland site. To accomplish this and to develop an understanding of how spatial gradients of contaminants affect the structure of benthic communities, the study design has to be robust enough in sample design and sensitive enough to differentiate real differences from natural variation. The approach used to support the original ecological risk assessment (SEH 1998a) was insufficient to draw any supportable conclusions regarding the relationship between the benthic macroinvertebrates community and environmental factors at Ashland, but those results can be used to develop the study design for a more comprehensive study.

As part of the study design, multiple sampling stations need to be established that can measure benthic community characteristics along gradients of contaminants as well as wood waste. In addition the study design needs to incorporate other factors that potentially affect benthic community composition, e.g., substrate type, depth, etc. It is unlikely this can be accomplished prior to evaluating the results of the surface sediment characterization discussed above and conducting a preliminary exploratory study to find sampling areas that meet the required study design parameters. In addition, reference stations representing different substrates also need to be identified. Reference sampling for background concentrations and effects should select sites that are similar to the Ashland site, but out of the influence of site contamination. Good guidelines for selecting reference locations are provided in the DOI NRDA regulations at 43 CFR Part 11.72.

The DQO process needs to be employed to understand the statistical power of the sampling design so that decision makers can recognize just how well the results can support risk management decisions In addition, the use of a number of multivariate techniques can be employed to identify environmental factors that affect the composition of benthic communities. These could include, for instance, various indices of "impairment" that can be differentiated by ordination analysis, for example as employed in the EPA Environmental Monitoring and Assessment Program for Estuaries, or selection of "pollution indicator species". The recent Baseline Ecological Risk Assessment for the Calcasieu Estuary employed a number of these numerical tools in their benthic macroinvertebrate survey to identify relationships among benthic community parameters and contaminants (CDM et. al 2001). The ARCS (Assessment and Remediation of Contaminated Sediment) guidance (USEPA 1994) also provides recommendations for benthic community analysis that include some of these approaches. Another approach is to compare benthic community parameters to abstract, "ideal" unimpacted communities using a multivariate approach (Reynoldson et al. 1995).

3.3.5 Wildlife Ingestion Study

Xcel Energy believes that the potential for impacts from sediment-associated contaminants to upper trophic levels is low and does not believe that this line of evidence will be influential in determining risk management options. However, if it is determined it is necessary to evaluate this assessment endpoint then more information is needed on potential sources of contaminants to wildlife ROCs in order to support a model of food chain transfer of contaminants associated with Ashland sediments. Data on tissue residue levels in fish are available (SEH 1998a), however, further information on contaminant levels in benthic invertebrate tissue levels, aquatic plants, if present, and contaminant levels in sediment of riparian zone should be considered to support such a model for an omnivorous aquatic-dependent bird, e.g. mallard duck. Xcel Energy believes that if there is any potential impact to aquatic-dependent wildlife, the greatest potential would be to riparian zone mammals. These include mink or birds such as the great blue heron, which feed primarily on fish and incidentally ingest sediment. Modeling of these riparian receptors should be sufficient to support risk management decision-making, so collection of tissue residue data from invertebrates and plants, if they are present, from the affect areas of the Ashland site may not be necessary.

3.3.6 Impacts to Fish

Xcel Energy believes that the fish tissue residue data collected to support the original ecological risk assessment (SEH 1998a) is adequate to support an analysis of food chain transfer. However, relating the results of this study to impacts to fish themselves is problematic since there isn't a credible database that associates tissue residue levels of most of the contaminants found at the Ashland site (i.e. PAHs and VOCs), with effects to fish. Toxicity studies are thus, arguably, the best method for evaluating direct effects and these have previously been conducted with the fathead minnow (SEH 2002).

The other line of evidence that can be used is to evaluate whether the levels of contaminants in Ashland sediment are sufficiently elevated and bioavailable to cause deformities, fin erosion of other histopathological effects to fish using the area. There are basically two approaches to evaluate this: One, comparison concentrations of contaminants found on site to benchmarks for these effects as derived from the literature; the other, to collect fish from the affected area, examine them and compare the results to fish from reference areas. The design of an evaluation such as this has to be carefully considered however since there may be factors other than site contaminants that cause these effects.

Xcel Energy does not believe that this line of evidence will be the most influential in determining risk management options since the ecological relevance of this endpoint is questionable. However, if other Interested Parties consider that this will be a critical line of evidence in risk management decision-making then a study should be developed to collect and examine resident fish.

3.3.7 Sediment Stability Analysis

The recently released draft "*Contaminated Sediment Remediation Guidance for Hazardous Waste Sites*" (USEPA 2002) includes fairly specific guidance for incorporating a consideration of sediment stability into the conceptual site model. A preliminary conceptual site model incorporating sediment stability and a recommended approach for evaluating sediment stability has been proposed following this guidance and is included as an Appendix to this document.

3.3.8 Evaluation of Wood Waste Impact

The potential affects of wood waste primarily relate to the fact that its presence on and in the sediment bed modifies potential benthic habitat, making it inhospitable to colonization through:

- Presence of saw dust or bark waste at the sediment surface presenting a physical barrier to colonization;
- Water soluble components of wood waste changing pH at sediment-water interface; and,
- Decreasing oxygen levels by increasing biological oxygen demand as wood decomposes.

Section 2.3.4 discusses the study design for a benthic community evaluation that is proposed for differentiating the effects of wood waste from that of contaminants.

3.4 Data Quality Objectives

3.4.1 Introduction of DQO Process

The Data Quality Objective (DQO) process is the USEPA approach for developing sampling designs for the collection of data that will be used in decision-making and has been recommended by USEPA ecological risk assessment guidance (USEPA 1997; 1998). This is the process for developing the detailed scope of the studies conducted to address the risk hypotheses formulated during the Baseline Problem Formulation (Section 1.0). This process uses systematic planning and hypothesis testing to differentiate between two or more clearly defined alternatives. A summary of the seven steps involved in the DQO process is presented in Table 6 (from USEPA 1998).

Table 6. Data Quality Objective Process.

DQO Step	Activity
Step 1. State the problem.	Review existing information to concisely describe the problem to be studied.
Step 2. Identify the decision.	Determine what questions the study will try to resolve and what actions may result.
Step 3. Identify inputs to the decision.	Identify information and measures needed to resolve the decision statement.
Step 4. Define study boundaries.	Specify time and spatial parameters and where and when data should be collected.
Step 5. Develop decision rule.	Define statistical parameter, action level, and logical basis for choosing alternatives.
Step 6. Specify tolerable limits on decision errors.	Define limits based on the consequences of an incorrect decision.
Step 7. Optimize the design.	Generate alternative data collection designs and choose most resource-effective design that meets all DQOs.

As described by USEPA (1997),

> "the DQO process represents a series of planning steps that can be employed throughout the development of the WP [Work Plan] and SAP [Sampling and Analysis Plan] to ensure that the type, quantity, and quality of environmental data to be collected during the ecological investigation are adequate to support the intended application. Problem formulation in Steps 3 and 4 is essentially the DQO process. By employing problem formulation and the DQO process, the investigator is able to define data requirements and error levels that are acceptable for the investigation prior to the collection of data. This approach helps ensure that results are appropriate and defensible for decision making. The specific goals of the general DQO process are to:

- Clarify the study objective and define the most appropriate types of data to collect;
- Determine the most appropriate field conditions under which to collect the data; and,
- Specify acceptable levels of decision errors that will be used as the basis for establishing the quantity and quality of data needed to support risk management decisions."

At this point in the Baseline Problem Formulation most of Steps 1 through 4 of the DQO process have been addressed. However, no "action level" or "decision statement" has been specifically formulated. The action level is the criterion for choosing among alternative actions (USEPA 2000). In the context of risk management it could be taken as the point at which the risk manager may decide to implement an active sediment remedy as opposed to allowing the sediment environment to naturally recover.

A decision statement is essentially an "If....then" statement that indicates at which point an alternative action will be implemented. As an example, a decision statement may state,

> "If the true mean concentration of bioaccumulative chemicals in a given species of fish or benthos in the impacted area exceeds a level (= the action level) demonstrated to cause adverse effects to populations of ecological receptors that prey on that benthic invertebrate or fish species and there is insufficient evidence that natural restoration will not change that situation in a reasonable amount of time then remedial measures such as removal or capping will be implemented in areas above the action level. However, if the true mean concentration of bioaccumulative chemicals in a given species of fish or benthos in the impacted area doesn't exceed a level demonstrated to cause adverse effects to populations of ecological receptors that prey on the fish or benthic invertebrate species then allow natural recovery to proceed and monitor."

Until the action level and decision statement are developed then Step 5, development of a decision rule, and Step 6, definition of tolerable limits on decision errors, can't be completed. USEPA (1998) comments on this,

> "The most important difference between problem formulation and the DQO process is the presence of a decision rule in a DQO that defines a benchmark for a management decision <u>before</u> (emphasis added) the risk assessment is completed. The decision rule step specifies the statistical parameter that characterizes the population, specifies the action level for the study, and combines outputs from the previous DQO steps into an "if . . . then" decision rule that defines conditions under which the decision maker will choose alternative options [....]. This approach provides the basis for establishing null and alternative hypotheses appropriate for statistical testing for

significance that can be effective in this application. While this approach is sometimes appropriate, only certain kinds of risk assessments are based on benchmark decisions."

The DQO process can be used for two "tiers" in problem formulation:

1) To support study design development. The DQO process can be used as the process for making decisions on sampling design and statistical protocol, e.g. determine how many samples must be collected in order to differentiate a 50% difference in a population parameter at some confidence level at a given Type I or Type II error rate.
2) To develop a decision framework for deciding whether the results of the risk assessment require that a specific remedy be implemented.

3.4.2 Use of DQO Process to Support Study Design Development for the Ashland Site

This use of the DQO process to develop study designs is relatively straightforward and won't be discussed in detail here. The DQO process can be used to optimize sampling design, decide upon how data will be analyzed, what levels of Type I and II error rates are tolerable, etc. The reader is directed to some of the referenced guidance or to the numerous texts on environmental sampling. It should be noted, however, that USEPA recommends that even these decisions should be made with the involvement of other Interested Parties. Clearly, all Interested Parties should be interested in the statistical reliability of the conclusions of any study conducted to support the lines of evidence used in the risk assessment.

3.4.3 Use of DQO Process to Support Risk Management Decision-Making for the Ashland Site

Notwithstanding USEPA's (1998) statement that the development of a decision rule or a decision framework up front is only appropriate in certain kinds of risk assessments, Xcel Energy believes that the Problem Formulation and DQO process should be used as the process for Interested Parties to develop an *a priori* decision framework that considers which remedies will be implemented based upon a range of risk assessment results. Engaging in discussions on what data are really important in deciding among alternative remedies quickly focuses risk managers on what information is needed to support risk management decisions. This also has the benefit of reducing the tendency to debate which remedies should be implemented once the risk assessment is completed. This, in turn, should reduce the potential for concluding at the end of the study that more and different data are needed before decisions can be made.

Although there has been considerable discussion over potential remedial approaches for the Ashland site involving a number of Interested Parties, there has been no explicit process defined to relate the results of the risk assessment to specific risk management decisions and remedial action outcomes (RAOs). As noted earlier in this problem formulation, it is important for Interested Parties to reach agreement early in the process on how the results of the risk assessment should be related to RAOs.

3.5 Weight of Evidence Evaluation

As indicated by USEPA (1997),

> "Confidence in the conclusions of a risk assessment may be increased by using several lines of evidence to interpret and compare risk estimates. These lines of evidence may be derived from different sources or by different techniques relevant to adverse effects on the assessment end points, such as quotient estimates, modeling results, or field observational studies. There are three principal categories of factors for risk assessors to consider when evaluating lines of evidence: (1) adequacy and quality of data, (2) degree and type of uncertainty associated with the evidence, and (3) relationship of the evidence to the risk assessment questions [...]. Data quality directly influences how confident risk assessors can be in the results of a study and conclusions they may draw from it. Specific concerns to consider for individual lines of evidence include whether the experimental design was appropriate for the questions posed in a particular study and whether data quality objectives were clear and adhered to. An evaluation of the scientific understanding of natural variability in the attributes of the ecological entities under consideration is important in determining whether there were sufficient data to satisfy the analyses chosen and to determine if the analyses were sufficiently sensitive and robust to identify stressor caused perturbations. Directly related to data quality issues is the evaluation of the relative uncertainties of each line of evidence [......]."
>
> Further, USEPA (1997) states that, "An interdisciplinary team including, but not limited to, biologists, ecologists, and environmental toxicologists, is needed to design and implement a successful risk assessment **<u>and to evaluate the weight of the evidence obtained</u>** (emphasis added) to reach conclusions about ecological risks."

Weighting of evidence should be used for two purposes:

1) Weighting of the value of the various measures or studies that are used to support a line of evidence, e.g, deciding which toxicological study is the most relevant in deriving a toxicological benchmark for use at a specific site; and,

2) Weighting of relative value of each line of evidence in determining what estimate of risk is most likely for site receptors given the results from that line of evidence.

The process of weighing various measures and studies or the relative value of each line of evidence may consider such factors as:

- Relevance of the study to the assessment endpoint;
- Strength of the exposure-response relationship;
- Appropriateness of the study temporal scope;
- Appropriateness of the study spatial scope;
- Quantity of data; and,
- Quality of data.

Xcel Energy recommends that as part of the problem formulation a process should be agreed upon <u>a priori</u> for determining how much weight should be assigned to each line of evidence used in this risk assessment. This process should take into consideration the scope of the risk management decision and the values of the various stakeholders. Relative weight of evidence of various lines of evidence can be agreed upon using a quantitative weighting approach (MADEP 1995) or can be consensually reached during Problem Formulation.

Xcel Energy proposes that the lines of evidence used in this risk assessment be accorded the following weight of evidence [numbered according to relative significance, with 1) having greater weight than 4)]:

1) Comparison of observed effects in the Ashland site benthic macroinvertebrate community characteristics to benthic macroinvertebrate community characteristics from reference area;

2) The results of bioassays conducted using standardized toxicity tests with sediments from the Ashland site and surrogate test organisms;

3) Comparison of tissue residue levels of contaminants in organisms collected from the Ashland site to tissue residue effects levels (e.g., critical body residue) for the same or similar species; and,

4) Comparison of site-specific media concentrations and/or ingested contaminant dose estimates (wildlife) to effects levels for the various ROCs.

Not all lines of evidence are necessarily used for each receptor group but when multiple lines of evidence are used the highest weight, i.e. the most important, should be accorded the results of site-specific studies. For instance, if there are conflicting results from the various lines of evidence, results from site-specific studies, e.g., from sediment bioassays using sediment from the Ashland site or comparison of the tissue residue levels of organisms collected at Ashland to tissue residue levels from similar species in reference areas, should be deemed more reliable for evaluating potential risk, then comparison to toxicological reference values for surrogate species. To the extent that additional lines of evidence are used for any of the assessment endpoints then Xcel Energy recommends a process be employed to reach consensus on the relative weight of evidence for these lines. As an example, if adequate information is available the Sediment Quality Triad (Chapman 1990) approach offers the benefit of precedent in interpreting multiple lines of evidence relating to sediment quality.

For those lines of evidence where hazard quotients were calculated, i.e., comparison to toxicological reference values or tissue residue effects levels, the risk hypothesis presented in Table 3 will be accepted if the hazard quotient was greater than one provided that the Type I and Type II error rates meet the DQO criteria. The acceptance of the risk hypothesis will be interpreted in this risk assessment to mean that the potential for adverse effects cannot be eliminated.

3.6 Risk Management Decision Framework

Xcel Energy recommends that Interested Parties develop a framework that integrates all lines of evidence used in this risk assessment and establishes an *a priori* risk-based management outcome for all or portions of the Ashland site that is based upon the outcome of studies conducted to support these lines of evidence. Xcel Energy proposes that the decision framework developed in Ontario Canada for decision-making at Great Lakes sediment sites (Grapentine et al. 2002) be used as a starting point for the Ashland site. This framework considers four primary lines of evidence: sediment chemistry, sediment toxicity, potential for biomagnification of site contaminants and effects to benthic macroinvertebrate community structure. Depending upon the strength of response under each line of evidence, the risk management measures recommended can vary from no action to sediment removal. Table 7 is an adaptation of their decision matrix and use of it is acknowledged.

In addition to the information in this matrix a consideration of sediment stability and an evaluation of sediment deposition rate needs to be factored into the decision process. Whether or not the sediment is stable in portions of the site and the degree to which sediment deposition can lead to natural recovery of all or portions of the site will be a determinant of whether the natural recovery option is viable. In that regard, it also should be recognized that the Ashland site is not a homogenous site and that some parts of the affected area likely should be managed differently than other parts. This aspect of site heterogeneity can be addressed by developing different decision matrices for different portions of the site.

3.7 Work Plan and Sampling and Analysis Plan

This is the culmination of Steps 3 and 4 of the Baseline Problem Formulation process. A detailed work plan will be developed for all studies that support the various lines of evidence using the QO process. This work plan will incorporate the results of the DQO process into the design of each study.

Table 7. Risk Management Decision Matrix.

Response – lines of evidence*						
Contaminant Concentration	Toxicity	Benthic Community	Biomagnification	Description of Current Status (surface sediment)	Interpretation	Management Option Recommended
-	-	-	-	Measured sediment contaminants not elevated above thresholds; no evidence of adverse biological effects.	Sediments do not present a risk.	No action.
+	-	-	-	Contaminants in sediment at elevated concentrations above thresholds, but not toxic; no evidence of adverse biological effects.	Contaminants do not present a risk.	No action. Monitoring recommended.
-	+	-	-	Measured sediment contaminants not present above thresholds; laboratory toxicity but no evidence of benthic community alteration in receiving environment; no biomagnification.	Potential for adverse effects.	No action. Monitoring recommended.
-	-	+	-	Evidence of non-sediment contaminant related stress (e.g., biotic or physical differences, water column conditions), or evidence of in situ stress due to unmeasured contaminant.	Sediments currently do not present a risk; however, other stressors are present.	Assess options for addressing presence of waste wood.

Response – lines of evidence*						
Contaminant Concentration	Toxicity	Benthic Community	Biomagnification	Description of Current Status (surface sediment)	Interpretation	Management Option Recommended
+	+	-	-	Contaminants in the sediment are toxic, but no evidence of benthic community alteration in receiving environment (e.g., due to acclimation / adaptation or insufficient stress to cause population-level responses or biomagnification).	Potential for adverse effects. Consider potential effects to unadapted communities in far field based on severity of the toxicity and assessment of site stability.	No action. Monitoring recommended.
-	+	+	-	Strong evidence of sediment related biological effects from unknown stressors (e.g., may be additive or synergistic effects of measured contaminants, or unmeasured contaminants or effects due to unmeasured habitat factor); no evidence for biomagnification by measured contaminants.	Adverse biological effects occurring.	Evaluate rate of natural recovery and spatial extent of impact. Select natural recovery alternative if area of impact is relatively limited and/or sedimentation rate is high. Monitor.

Response – lines of evidence*						
Contaminant Concentration	**Toxicity**	**Benthic Community**	**Biomagnification**	**Description of Current Status (surface sediment)**	**Interpretation**	**Management Option Recommended**
-	-	+	+	Evidence of either increased biological bioavailability or non-sediment contaminant related stress (e.g., biotic or physical differences, water column conditions, unknown habitat factors); potential for risk at higher trophic levels but source(s) unclear.	Sediments currently may not pose a risk, but biomagnification occurring. Evaluate alternative explanations for benthic community impacts. Is toxicity chronic or is impact to benthic community related to other factors.	If risk from biomagnification is significant, dredge or cap sufficient portions of site to eliminate risk. If biomagnification risk is minor, evaluate rate of natural recovery and spatial extent of impact. Select natural recovery alternative if area of impact is relatively limited and/or sedimentation rate is high. Monitor.
+	-	+	-	Benthic community alteration may be due to sediment contaminants or other stressors; no toxicity, or effects may be too chronic/long term for detection by toxicity tests; no biomagnification.	Adverse effects occurring but cause(s) unknown.	Evaluate further. Monitor.

Response – lines of evidence*						
Contaminant Concentration	Toxicity	Benthic Community	Biomagnification	Description of Current Status (surface sediment)	Interpretation	Management Option Recommended
+	-	-	+	Potential for risk at higher trophic levels related to sediment contamination above thresholds.	Unacceptable risk of biomagnification.	If risk from biomagnification is significant, dredge or cap sufficient portions of site to eliminate risk. If biomagnification risk is minor, evaluate rate of natural recovery and spatial extent of impact. Select natural recovery alternative if area of impact is relatively limited and/or sedimentation rate is high. Monitor.
-	+	-	+	Measured sediment contaminants not present above thresholds; laboratory toxicity not expressed in receiving environment (e.g., due to acclimation / adaptation or insufficient stress to cause population-level responses); possible additive/synergistic effects from low levels of contaminant mixtures; potential for risk at higher trophic levels but sources unclear.	Potential for adverse effects to benthos; unacceptable risk of biomagnification.	If risk from biomagnification is significant, dredge or cap sufficient portions of site to eliminate risk. If biomagnification risk is minor, evaluate rate of natural recovery and spatial extent of impact. Select natural recovery alternative if area of impact is relatively limited and/or sedimentation rate is high. Monitor.

Response – lines of evidence*						
Contaminant Concentration	**Toxicity**	**Benthic Community**	**Biomagnification**	**Description of Current Status (surface sediment)**	**Interpretation**	**Management Option Recommended**
+	+	+	-	Elevated sediment contaminants likely causing adverse biological effects.	Sufficient evidence for unacceptable risk from sediment contamination.	Evaluate rate of natural recovery and spatial extent of impact. Select natural recovery alternative if area of impact is relatively limited and/or sedimentation rate is high. Monitor.
+	+	-	+	Measured sediment contaminants present above thresholds; laboratory toxicity not expressed in receiving environment (e.g., due to acclimation / adaptation or insufficient stress to cause population-level responses); potential for risk at higher trophic levels. Responses may be due to two or more separate mechanisms or stressors.	Potential for adverse effects to benthos; risk of biomagnification.	If risk from biomagnification is significant, dredge or cap sufficient portions of site to eliminate risk. If biomagnification risk is minor, evaluate rate of natural recovery and spatial extent of impact. Select natural recovery alternative if area of impact is relatively limited and/or sedimentation rate is high. Monitor.

Response – lines of evidence*						
Contaminant Concentration	**Toxicity**	**Benthic Community**	**Biomagnification**	**Description of Current Status (surface sediment)**	**Interpretation**	**Management Option Recommended**
+	-	+	+	Measured sediment contaminants present above thresholds; benthic community alteration may be due to sediment contaminants or other stressors; no toxicity or effects may due to chronic/long term toxic action; potential for risk at higher trophic levels. Responses may be due to two or more separate mechanisms or stressors.	Adverse effects to benthos; unacceptable risk of biomagnification.	If risk from biomagnification is significant, dredge or cap sufficient portions of site to eliminate risk. If biomagnification risk is minor, evaluate rate of natural recovery and spatial extent of impact. Select natural recovery alternative if area of impact is relatively limited and/or sedimentation rate is high. Monitor.
-	+	+	+	Adverse biological effects and biomagnification related to unmeasured sediment contaminants, or synergistic / additive effects of measured sediment contaminants.	Adverse effects to benthos; unacceptable risk of biomagnification.	If risk from biomagnification is significant, dredge or cap sufficient portions of site to eliminate risk. If biomagnification risk is minor, evaluate rate of natural recovery and spatial extent of impact. Select natural recovery alternative if area of impact is relatively limited and/or sedimentation rate is high. Monitor.

Response – lines of evidence*						
Contaminant Concentration	Toxicity	Benthic Community	Biomagnification	Description of Current Status (surface sediment)	Interpretation	Management Option Recommended
+	+	+	+	Elevated sediment contaminants above thresholds likely causing adverse biological effects and biomagnification.	Sufficient evidence for unacceptable risk from sediment contamination.	If risk from biomagnification is significant, dredge or cap sufficient portions of site to eliminate risk. If biomagnification risk is minor, evaluate rate of natural recovery and spatial extent of impact. Select natural recovery alternative if area of impact is relatively limited and/or sedimentation rate is high. Monitor.

"-" denotes that, for the decision element, there is no indication of contamination or an adverse biological condition.
"+" = denotes that, for the decision element, there is a clear indication of contamination or an adverse biological condition. The certainty associated with a "+" response can range in strength. Note that each decision element is considered to be reasonably definitive (e.g., risk has been shown, not just hazard; there are no false positives in the toxicity tests, etc.).

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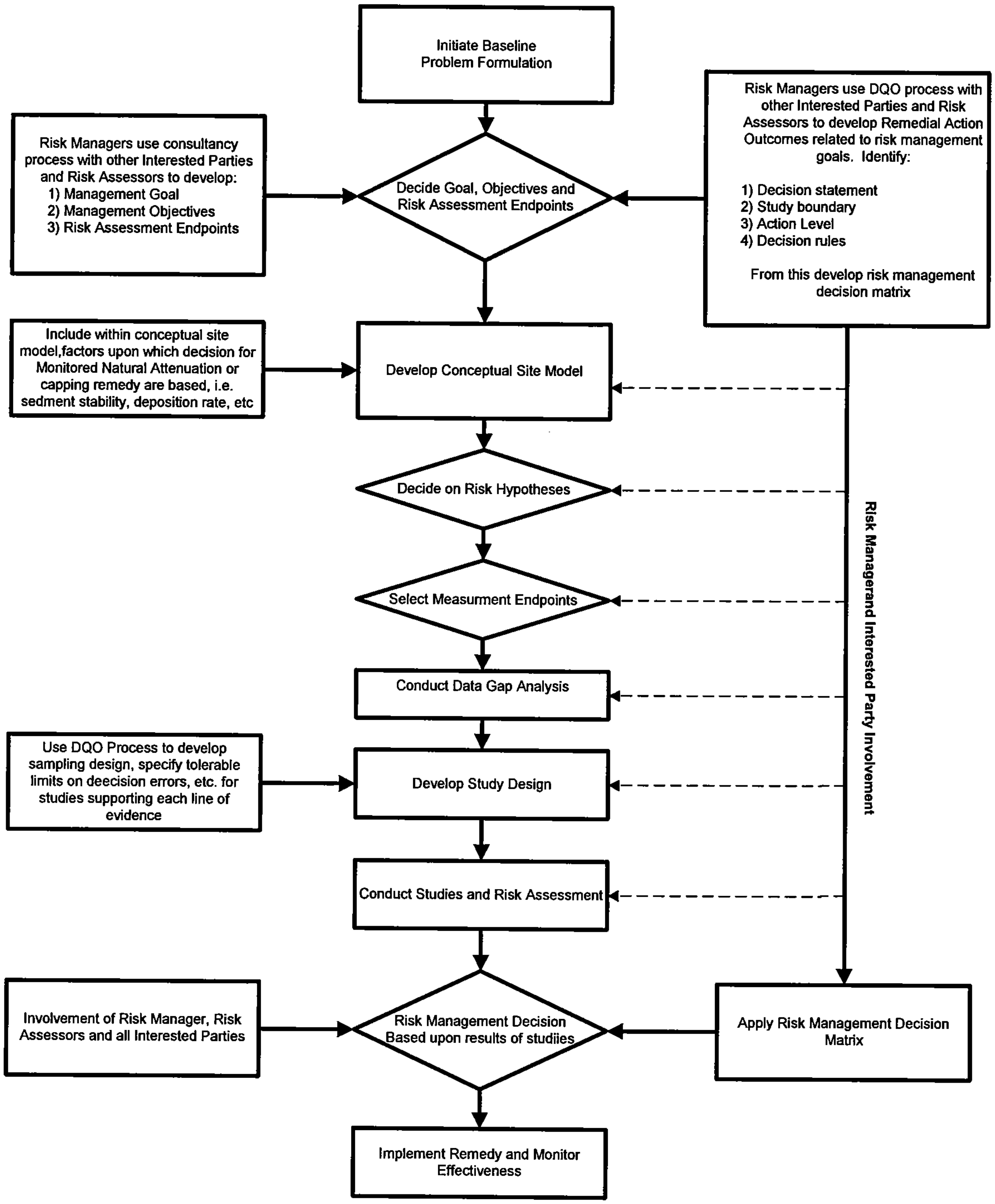


Figure 1. Flow chart illustrating integration of risk assessment problem formulation, DQO process and contaminanted sedment management guidance.

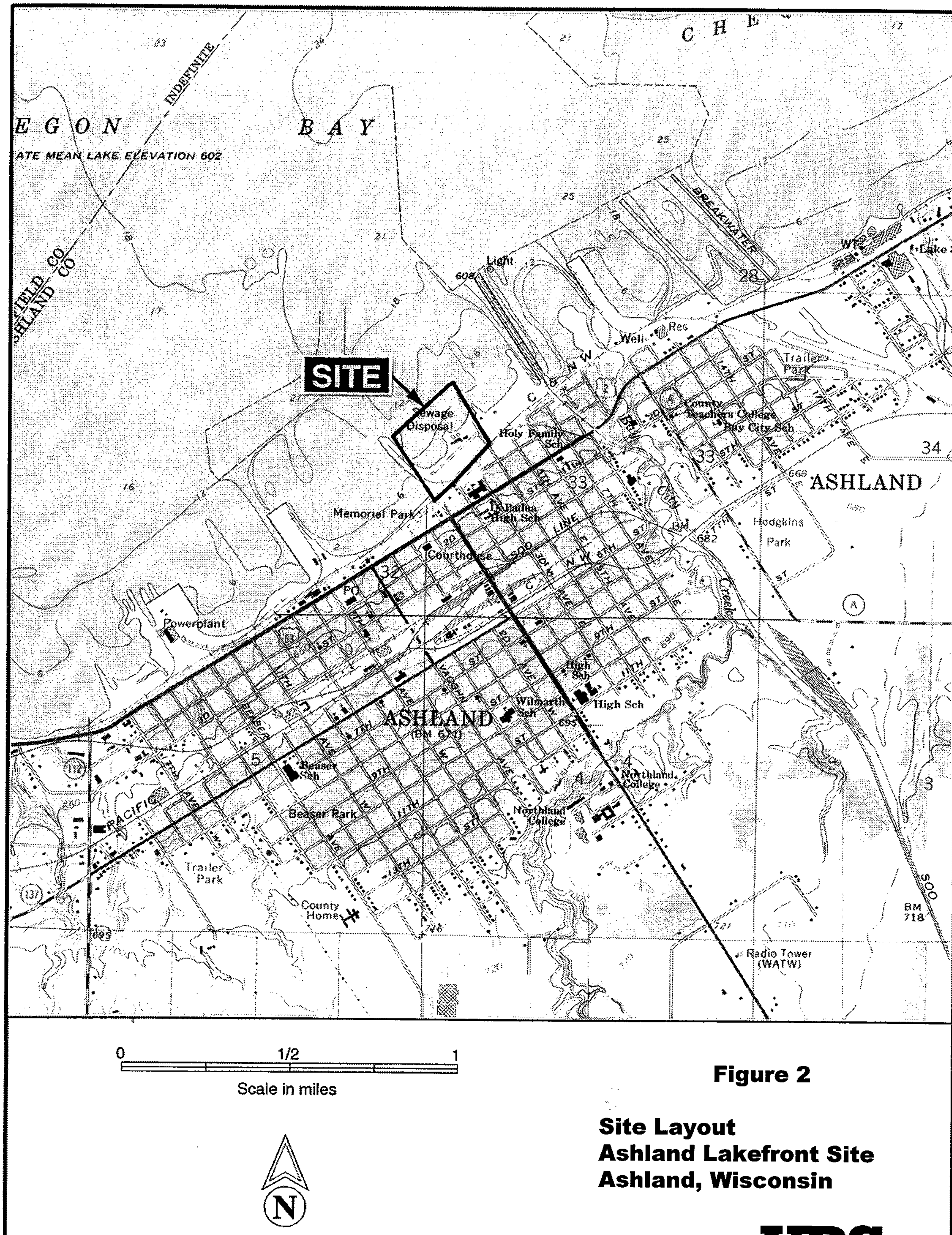
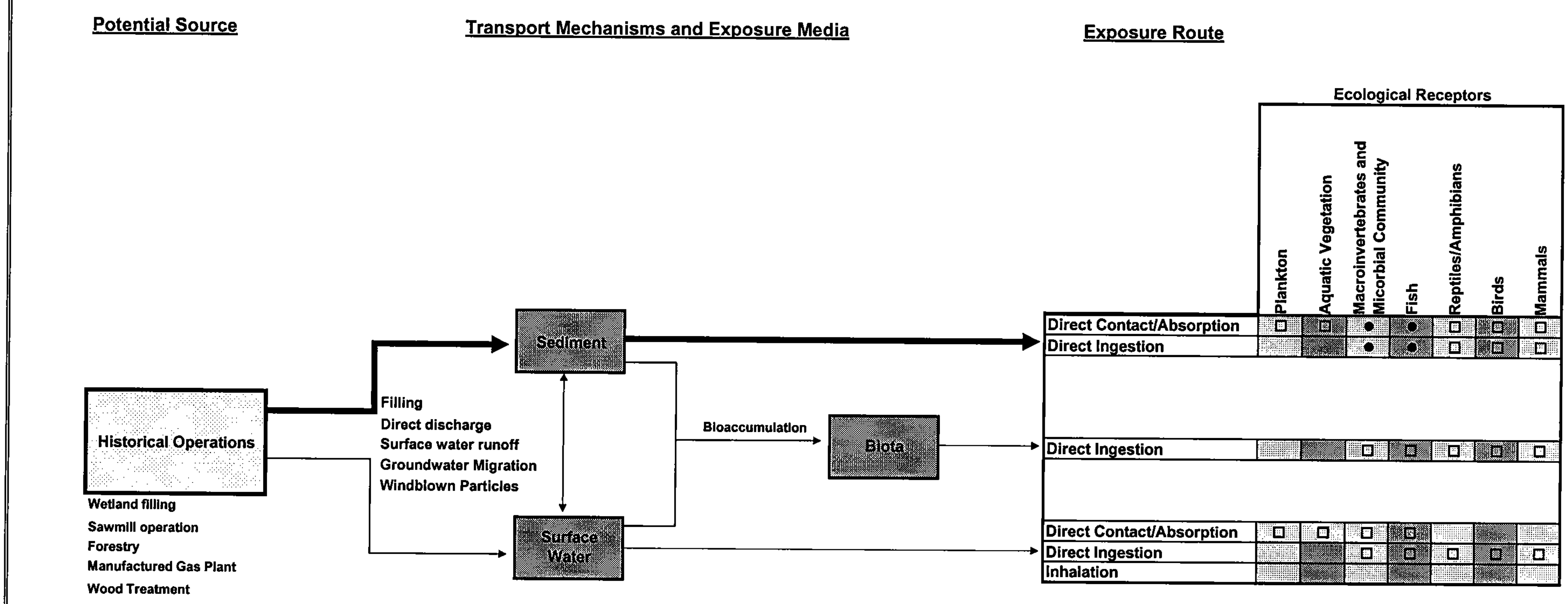


Figure 2

Site Layout
Ashland Lakefront Site
Ashland, Wisconsin

URS

REFERENCE: USGS 7.5 Minute Series Topographic Maps: "Ashland East, Wisconsin" and "Ashland West, Wisconsin" Quadrangles, Photorevised 1975.



NOTE:
● = Potentially Important Exposure Pathway
□ = Minor Exposure Pathway
Blank = Incomplete Exposure Pathway

FIGURE 3

CONCEPTUAL SITE MODEL
AQUATIC EXPOSURE PATHWAYS
Ashland Lakefront Site
Ashland, Wisconsin

APPENDIX

Approach to Evaluating Sediment Stability at the Ashland Lakefront Site

1.1 Introduction

This preliminary work plan has been developed to evaluate sediment stability in the sediment Operable Unit at the Ashland Lakefront Site. The objective of this proposed study is to determine the potential for existing sediment to be eroded or buried. If the potential for erosion and deposition are determined to exist, net sediment transport rates to and from the study area will be estimated. The results of the study will also be used to estimate the amount of associated sediment contamination that is potentially re-exposed and transported and the rate at which the contaminated sediments are being buried due to natural sediment deposition.

The proposed work plan will employ both quantitative (i.e. modeling) and empirical techniques to determine sediment stability as recommended in the EPA Contaminated Sediment Remediation Guidance for Hazardous Waste Sites (USEPA 2002). Both of these approaches will be implemented in a screening level phase, and if the results of the screening level analysis indicate further investigation is warranted, a detailed analysis will be performed. Although the two approaches may involve different methods, they will be used jointly to draw conclusions about sediment stability at the Site.

1.2 Quantitative Approach Work Plan

The screening level quantitative approach consists of estimating the potential for erosion of existing sediments using process-based quantitative models of erosion and deposition. The sediment Operable Unit is in relatively shallow water and sheltered by marina structures that limit both wave propagation and circulation. However, tidal flows, waves generated by winds from the northeast and storm-generated currents potentially could provide hydrodynamic forcing to erode the contaminated sediments in the Site area. Therefore a wave/current interaction model, based on the Glenn and Grant (1987) and recent Madison (1999) methods will be implemented. Recent modifications to these approaches have been developed to account for bedform (ripples, sand waves etc.) evolution and migration and their effect on hydrodynamic response and sediment transport. These models require wave period, height and direction and current speed and direction as well as sediment characteristics as input.

For sediment stability analysis of contaminated sediments, it is necessary to consider the full range of forcing events that may lead to erosion. A single event may produce a small amount of erosion and transport, but when many events are integrated over extended time periods, they also could potentially lead to significant changes. Thus a statistical representation of forcing conditions for the area will be developed and the erosion analysis will be conducted for a range of conditions. The results will be combined to estimate the long-term potential for erosion.

To develop model inputs, a literature search will be conducted. The necessary information on measurements of waves and currents at the Site and in adjacent areas will be identified if available. If these data are not readily available, meteorological data will be used to estimate wave and current conditions in the area.

Data on sediment characteristics required for the quantitative approach depend on whether the sediments exhibit cohesive behavior. For non-cohesive sediments, which are generally in the medium silt to sand size range, grain size curves, mineralogy and bulk density will suffice. These parameters, along with data from numerous laboratory studies of erosion, are sufficient to characterize the erosion characteristics of the non-cohesive sediment. However, if there is a significant amount of fine silt or clay size particles (> 10%) or there is sufficient organic material in the sediments, the sediments may be classified as cohesive.

A review of existing data available in previous risk assessments (SEH 1998; 2002) indicates that cohesive characteristics may occur. The erosion characteristics of cohesive sediments are very site-specific, and there is very little guidance available in the literature for developing the characteristics from bulk sediment properties. The best method to characterize the erosion potential of Site sediments is to use laboratory testing of Site sediment samples or in-situ erosion testing. Both of these methods will be pursued if warranted by the study.

The presence of wood chips may alter the erosion characteristics of the sediments and create conditions that are unlikely to have been studied previously. Therefore a diligent effort to obtain site-specific estimates will be made. It is noted that standard erosion testing may be difficult for contaminated sediments due to health and safety issues. If no laboratory can be found to handle the sediments then we will rely on data from the literature and best engineering/scientific judgement to develop the erosion parameters.

The results of this quantitative analysis will be combined and evaluated in conjunction with conclusions drawn from the empirical analysis described below, which also will be conducted simultaneously as part of the screening level phase. Quantitative approaches are known to be sensitive to model parameter values as well as dependent on site-specific data, which is often difficult to obtain. The empirical study will provide insight into the site-specific sediment transport processes and yield quantitative bounds that can be used to constrain and guide the quantitative analysis.

1.3 Empirical Approach Work Plan

The empirical analysis will develop a conceptual model that best characterizes sediment stability at the Site. The conceptual model developed as part of this empirical analysis will attempt to describe the historical development of the existing contaminated sediments by considering various possible sediment and contaminant transport mechanisms and pathways. It may also be used to estimate historic site-specific transport rates and provide a basis for estimating future rates of sediment erosion and deposition.

The empirical analysis will rely mostly on analysis of existing data such as the sediment borings, vertical profiles of contamination, sediment deposition rates, physical sediment characteristics as well as other Site data on land use characteristics, regional hydrography, river sediment loading rates, and contaminant behavior in sediment, soils and water.

Contamination in the Ashland sediments appears to be mainly confined to a sediment layer extending a few hundred feet from the shoreline. The layer has a maximum thickness near the shoreline, typically 3 to 4 feet, and tapers off in the offshore direction. The layer is characterized by the presence of wood chips and most of the contamination is confined to this layer, although some contamination appears to exist just below the wood chip layer. The wood chips apparently were derived from local material, either from the sawmill or directly from logs floated and rafted into the Ashland area.

Based upon a review of existing data, there are at least two possible conceptual models that explain the current contaminant distribution at the Site contamination. These models are preliminary, but serve as a starting point for the empirical analysis.

In the first conceptual model, it is noted that much of the existing shoreline and some of the marina structures were created by back-filling soil into the bay. It is possible that the process of back-filling, which is assumed to have occurred episodically between 120 and 60 years ago, created most of the contaminated wood-chip laden sediment layer. The backfill material, which was likely generated from the bluff and surrounding area, contained wood processing wastes as well as contamination from facilities operating in the area. Xcel Energy has produced documentation that it believes indicates the PAH contamination measured in the sediments has been generated from multiple sources (e.g., wood treatment at the former Schroeder Lumber Company and the former manufactured gas plant). As this material was transported to the harbor, some of it escaped into the surface water and settled out in the near shore area. The shape of the wood-chip layer, thick near the shoreline and tapering offshore, is consistent with this view.

In this interpretation, most of the contamination was derived from existing soil and surface contamination associated with the back-fill material. It is also possible that sediments associated with surface runoff and groundwater transport contributed to the development of the deposit. Both contaminated and uncontaminated sediments reached the Site from rainfall induced surface runoff originating in the watershed adjacent to the Site. It is alternatively possible that contaminated surface runoff mixed with re-suspended sediments and contributed to the contaminated sediment layer as it evolved. However, it is likely that these processes only played a secondary role relative to the contamination derived from the back-filling.

If this interpretation is substantiated, it may imply that the sediments are fairly stable. The contaminated layer would not require historic or ongoing active sediment transport to have developed, and therefore it is possible the transport is low and insignificant in the area and sediments are relatively stable.

Additional data will help quantify and verify this conceptual model. One or more soil borings taken just inland of the shoreline will be analyzed for sediment content. If a wood-chip-laden layer is also present in the borings, and at elevations consistent with the offshore borings, then it is evidence that the contaminated layer was created predominantly by back-filling activities. Additionally, higher resolution vertically-stratified profiles of key contaminants in a few select sediment cores could provide additional information for evaluating historic and current transport processes and rates. Finally, if analysis of the inland soil borings do not provide sufficient information, it may be necessary to conduct age dating (most likely Pb_{210}) to help quantify transport rates.

An alternate conceptual model for the evolution of the contaminated sediment layer is based on regional sediment transport patterns. In this interpretation, much of the sediment that comprises the layer may have originated up and down shore of the Site. Sediment was (and possibly still is) transported via waves and currents all along the shoreline and during high energy events and the sediment that makes its way into the Site will deposit during the waning phase of the event. Although the sediment may have been contaminant-free at its origin, it likely mixed with contaminated runoff and/or contaminated sediments from the watershed adjacent to the Site, and then deposited at the Site.

It is known that logging was active during the last 120 years in the region. Logging could have provided a steady source of sediment to the harbour area since logging activities are known to increase soil erosion and provide additional source of sediments to rivers. A review of stream and river networks in the area show two drainage basins, one to the east and a larger one to the west. These rivers may have carried relatively large sediment loads to Chequamegon Bay, some of which eventually were deposited at the Site. It is likely that the current sediment load has been reduced relative to loads that occurred during the period of relatively uncontrolled historic logging activities, due either to reduced logging activities and/or improvement in logging procedures.

In this conceptual model, the evolution of the contaminated sediment layer occurred fairly continuously, due primarily to the sediment loads associated with the regional logging industry. In a large-scale long-term view, for the period of 120 to 60 years ago, the Bay was unable to flush the anthropogenic source of sediments at the rate that they were supplied. The hydrodynamic forces may have been able to remove some of the additional load to deeper water but not all of it. Thus the sediments began to accumulate along the shoreline as well as in deeper waters in the Bay. In terms of sediment balance, there was net sediment input into the Bay. If the logging industry based sediment load has actually been reduced, then the sediment balance near the coastline may begin shifting from depositional to erosional. However, the erosion rates may be relatively low, due to the relatively low hydrodynamic energy and limited fetch lengths in the area. In either case, quantification of this conceptual model will determine net erosion or depositional rates.

Additional data needed to quantify this conceptual model is similar to that needed for the previous one, except that the upland soil borings will not be needed. The age dating profiles from sediment borings could have one of two characters, either ages fairly uniform throughout the layer, which indicates that back-filling may be the dominant process or they may indicate a more gradual development, consistent with the regional sediment transport interpretation. Similar conclusions can be drawn from the structure of high-resolution (i.e. 2 cm) contaminant vertically-stratified profiles in key sediment cores.

The preliminary analysis of empirical data described above is meant to be a starting point for the screening level phase. Clearly, additional data exists which has not been included in developing these models. The work plan for the empirical analysis consists of further development of these models, inclusion of other data sources and results from the proposed additional data collection, as well as considering alternate hypotheses individually or in combination. Other data that could be helpful are USGS river flow and sediment load data, either from adjacent watersheds or nearby watersheds, historic and current land use maps, detailed operational data for the Site and adjacent areas (including regional logging practices), meteorological data, and oceanographic data for Chequamegon Bay.

At the conclusion of the screening level phase, results from the quantitative and empirical approaches will be combined to make an assessment of sediment stability at the Site.

If the screening level analysis does not produce conclusive estimates of the sediment stability, an additional phase will be conducted which will be based on hydrodynamic and circulation modeling. The model will be used to estimate sediment deposition and erosion in and around the Site. This analysis will involve additional data collection for model calibration and forcing. The model will be either a 2 or 3D numerical model, consistent with the US Army Corp of Engineers Surface Water Modeling System ADCIRC and M2D models. The data collection and modeling plan will be developed at the time the detailed modeling is determined necessary (i.e., at the end of the screening level phase). Data and information obtained in the screening level phase will be used to guide the development of the data collection and modeling phase.

1.4 References

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